MARYLAND BIOLOGICAL STREAM SURVEY 2000-2004

Volume XIV

Stressors Affecting Maryland Streams

MARYLAND DEPARTMENT OF NATURAL RESOURCES

CHESAPEAKE BAY AND WATERSHED PROGRAMS
MONITORING AND NON-TIDAL ASSESSMENT
CBWP-MANTA-EA-05-11

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Secretary

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FOREWORD

This report, *Maryland Biological Stream Survey 2000-2004 Volume14: Stressors Affecting Maryland Streams*, was prepared by Versar, Inc., as a combined effort of several authors from Versar, University of Maryland, and Maryland DNR, for the Maryland Department of Natural Resources' Monitoring and Non-Tidal Assessment Division. It was supported by Maryland's Power Plant Research Program (Contract No. K00B0200109 to Versar, Inc.).

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ABSTRACT

This volume uses the data collected by the MBSS since 1994 to identify the stressors (e.g., individual pollutants, physical habitat changes, invasive species, and general factors such as land use) that are affecting Maryland's streams. To the surprise of no one, a large proportion of our streams are in poor condition and many more are in worse condition than we desire. The first step in "fixing" these streams is determining why they are "broken." Identifying stressors is critical to meeting Clean Water Act mandates and developing management actions that can restore or protect the desired condition of streams. Stressor identification, or the diagnosis of stream problems, is an emerging field that draws on the approaches of traditional risk assessment while using new metrics derived from more sophisticated monitoring data. Relative risk assessment and cumulative impact analyses are another approach useful for setting management priorities. This volume uses both approaches to investigate which stressors are responsible for degradation of Maryland streams.

Stressors can be organized according to the five major determinants of biological integrity in aquatic ecosystems: water chemistry, energy source, habitat structure, flow regime, and biotic interactions. Water chemistry comprises acidity, dissolved oxygen, and contaminants. Energy source describes the size, abundance, and nutritional quality of food from both primary production and allochthonous inputs. Habitat structure encompasses physical features such as water depth, current velocity, substrate composition, and morphology of the stream channel. Flow regime refers to seasonal, annual, and altered patterns in water quantity and delivery. Biotic interactions include competition, predation, and parasitism, from both native and introduced species. The MBSS directly measures many of these stressors and ancillary information, such as land use, can be used to evaluate others. Some stressors, such as pesticides, currently are not considered in MBSS analyses. This volume includes detailed analysis of five important stressor categories: acidification, nutrients, physical habitat, biotic interactions, and land use. It also discusses the relative contributions of each stressor and their cumulative impact on stream resources. Lastly, it provides basin and site examples of stressor identification.

Important results include the strong effect that acidification (especially low acid neutralizing capacity, ANC) has on both fish and benthic macroinvertebrate communities. While acid mine drainage is among the most severe stressors (producing a strong effect when present), the extent of streams affected by AMD (1% of all stream miles) is small compared to other stressors, including acidic deposition. Aquatic non-natives and invasive plants are the stressors affecting the greatest number of stream miles statewide (more than 50%). Nutrient pollution also affects Maryland streams as evidenced by the strong relationship between sensitive benthic taxa (Ephemeroptera, Plecoptera, and Trichoptera) and the ratio of total nitrogen (TN) to total phosphorus (TP). The percentage of agricultural land use is a good predictor of nitrate levels in streams. Degradation of instream physical habitat is the stressor most often resulting in the loss of individual fish species. Urban land use and its concomitant impervious surfaces strongly affect the fish, benthic, and salamander communities in streams to the point that other stressors are obscured and management solutions may be limited.

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14.1 BACKGROUND

The Maryland Biological Stream Survey (MBSS) provides State agencies and the public with a comprehensive assessment of the condition of Maryland's streams. To the surprise of no one, a large proportion of our streams are in poor condition and many more are in worse condition than we desire. The first step in "fixing" these streams is determining why they are "broken."

Identifying stressors is critical to the development of management actions that can restore or protect the desired condition of streams. To implement Section 303(d) of the Clean Water Act, the Maryland Department of the Environment (MDE) must identify stressors ("pollutants") for impaired waters so that Total Maximum Daily Loads (TMDLs) can be developed. These TMDLs are the State's mechanism for maintaining water quality standards. Stressor identification also supports Clean Water Act implementation through 305(b) Water Quality Reports, 319 Nonpoint Source Control, 402 Point Source Permitting, and 401 Water Quality Certifications. In addition, the Maryland Department of Natural Resources (DNR) and other agencies maintain active restoration programs for Maryland's streams. Programs, such as Watershed Restoration Action Strategies (WRAS) use MBSS and other data (e.g., Stream Corridor Assessment) to identify stressors. The Chesapeake Bay Program, as well as other governmental and private programs, also must identify stressors to implement their restoration initiatives. This volume uses the data collected by the MBSS since 1994 to summarize the severity and geographic extent of stressors (individual pollutants, physical habitat changes, and general factors such as land use) that are affecting Maryland's streams. It does not provide a complete characteristic of stressors in Maryland (additional analyses will be conducted in the future) nor is it a formal stressor identification as described by EPA (see below) to support regulatory decisions.

14.1.1 Stressor Identification

Stressor identification, or the diagnosis of stream problems, is an emerging field that draws on the approaches of traditional risk assessment while using new metrics derived from more sophisticated monitoring data. The U.S. Environmental Protection Agency (2000a,b,c) has published a stressor identification process that includes three steps (Figure 14-1):

- Listing candidate causes
- Analyzing new and previously existing data to generate evidence for each candidate cause
- Producing a causal characterization using the evidence generated to draw conclusions about the stressors that are most likely to have caused the impairment.

Critical to characterizing causes are the three approaches of eliminate, diagnose, and strength of evidence.

Stressors can be organized according to the five major determinants of biological integrity in aquatic ecosystems (Figure 14-2): water chemistry, energy source, habitat structure, flow regime, and biotic interactions (Angermeier and Karr 1994, Karr and Chu 1998). Water chemistry comprises acidity, dissolved oxygen, and contaminants. Energy source describes the size, abundance, and nutritional quality of food from both primary production and allochthonous inputs. Habitat structure encompasses physical features such as water depth, current velocity, substrate composition, and morphology of the stream channel. Flow regime refers to seasonal, annual, and altered patterns in water quantity and delivery. Biotic interactions include competition, predation, and parasitism, from both native and introduced species.

The MDE is considering individual candidate causes as a means of addressing specific problems within these five factors:

- Chemical
  - Chemical toxicity
  - Low dissolved oxygen
  - pH
- Energy source
  - Increased primary production
  - Decreased allochthonous input
- Physical habitat
  - Sediment
  - Channel modification
  - Temperature
- Flow regime
  - High discharge
  - Low discharge
- Biotic interactions
  - Exotics
  - Pathogens
  - Exploitation

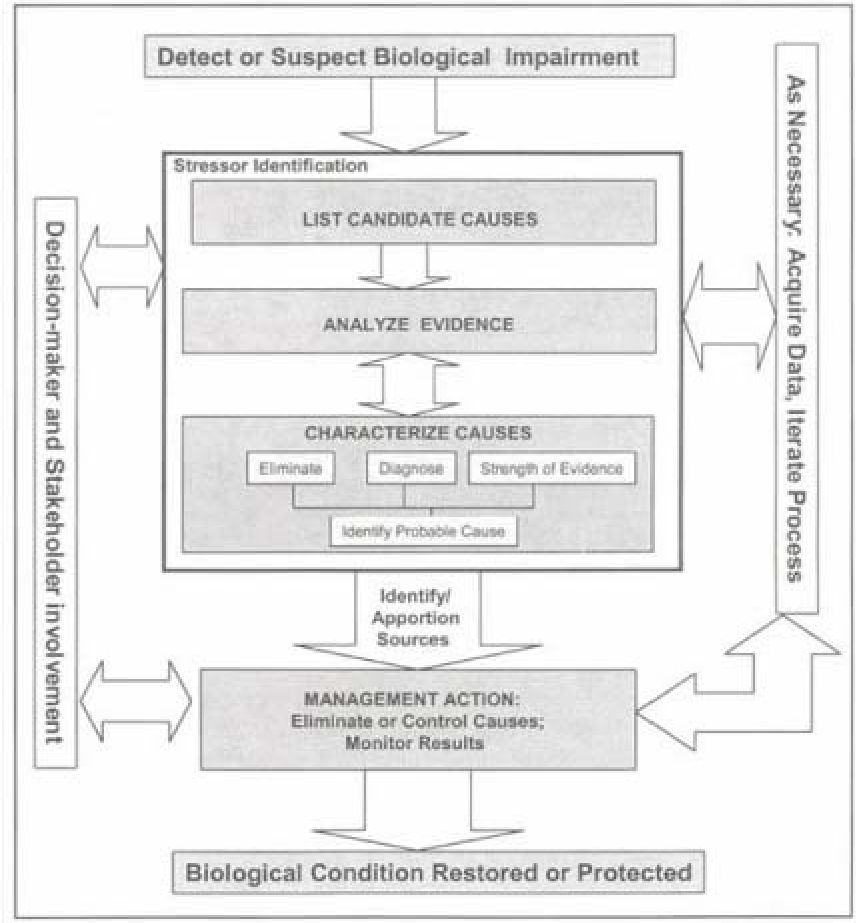


Figure 14-1. The management context of the stressor identification process. (The stressor identification process is shown in the center box with bold line. Stressor identification is initiated with the detection of a biological impairment Decision-maker and stakeholder involvement is particularly important in defining the scope of the investigation and listing candidate causes. Data can be acquired at any time during the process. The accurate characterization of the probable cause allows managers to identify appropriate management action to restore or protect biological condition.)

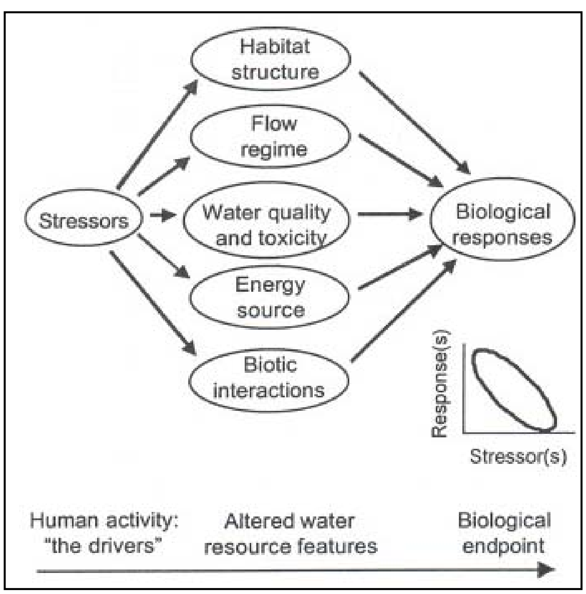


Figure 14-2. Linkages from human activity (the stressors or drivers of system change) through the five major water resource features altered by human activity, to the biological responses producing ambient condition, i.e., the biological endpoints of primary interest in biological assessment programs. This model illustrates the multiple causes of water resource changes associated with human activity. Insert illustrates the relationship between stressor dose and the gradient of biological responses that signal a good biological metric. (Source: Karr and Yoder 2005).

Stressor identification requires that each candidate cause have a conceptual model that shows the relevant cause-and-effect relationships. The MDE has developed preliminary conceptual models of the 13 candidate causes, though these models are still evolving. Compiling and interpreting evidence on which cause is affecting stream condition is the most critical part of stressor identification. This is analogous to diagnosing the illness of a patient. The accuracy and precision of the diagnosis depends on the relevance and sophistication of the evidence that can be obtained, including whether the evidence describes the effect directly, the exposure to a stressor, the cause of that stressor, or the original source of the stressor.

The MBSS collects a core of site data on biological assemblages, physical habitat, and water chemistry in all of Maryland's watershed basins (Figure 14-3). This information is supplemented with land use and other area-wide data from many sources. This combination of data can be used to identify specific stressors at individual sites and predominant stressors at the basin, tributary basin, or statewide level. In some cases, stressor identification in streams can be straightforward, as when pH levels are below thresholds known to adversely affect fish survival. In other cases, the evidence is a mixture of habitat changes (embedded substrate) and indirect causes (stormwater runoff from impervious surfaces). A promising method of stressor identification is the use of biological data when consistent thresholds of tolerance have been determined. The large number of sites sampled by the MBSS across the full range of human disturbance allowed Stranko et al. (2005a) to develop a Prediction and Diagnosis Model that can be used to identify likely stressors when fish species are absent. All of these methods are used in this volume to identify which stressors are affecting Maryland streams, and to characterize their severity and geographic extent.

14.1.2 What's in This Volume?

This volume attempts to look broadly at what stressors may be affecting Maryland streams, but it cannot be comprehensive because evidence for some stressors are not available. For example, migration barriers to fish have an important affect on fish communities but many such barriers cannot be identified during site visits or from currently available data sources. Similarly, many upstream effects are also not discernible at the site level nor can they be determined with large-scale ancillary data. Local assessment data (such as Stream Corridor Assessment data) are very helpful at fine scales, but will be introduced here only. The range of evidence that is available from the MBSS and separate landscape data are powerful, however, and will be used in the sections that follow.

To limit the size and complexity of this volume and increase readability, all methods used to prepare and analyze data for this volume are presented in 2000-2004 Maryland Biological Stream Survey Volume 6: Laboratory, Field, and Analytical Methods. This volume can be downloaded from http://www.dnr.Maryland.gov/streams/pubs.

Another caution about stressor identification is the difficulty in determining when individual stressors interact to produce synergistic (or antagonistic) effects on biota. To some extent, land use reflects a suite of stressors

STREAM CORRIDOR ASSESSMENTS

The MBSS provides excellent coverage of the State at the scale of the Maryland 8-digit basins (approximately 50 mi^2). The mean IBIs for fish and benthic macroinvertebrate assemblages (and the proportion of stream miles in each IBI class) provide robust measures of the cumulative stress on biological communities, at this scale. In addition, stressor and stressor-surrogate variables sampled at MBSS sites provide areawide estimates of the extent and severity of water quality and physical habitat conditions. MBSS data do not, however, provide coverage of stressor presence at the next larger stream reach scale. For example, when evaluating the stressors potentially affecting biological condition at an individual MBSS site, the presence of an adequate riparian buffer along the 75-m sample segment could be misleading if the riparian buffer has been removed along the entire reach upstream of the site. Stressor identification in Maryland streams would be greatly enhanced if data on the reach level could be combined with MBSS data collected at the segment level. Fortunately, Maryland DNR is conducting reach-level Stream Corridor Assessments (SCAs) as part of the State's Watershed Restoration Action Strategies (WRASs) in selected 8-digit basins (Yetman 2001). Analysis is underway to evaluate how MBSS and SCA data might be combined for improved stressor identification.

What is SCA? SCA data are intensive, covering the entire stream network of selected 8-digit basins through "stream walks," which inventory each individual problem site along a stream, and "representative sites," which document the instream and riparian habitat conditions along small stretches of a stream (approximately 300 feet in length). These habitat assessments are based on an array of habitat metrics similar to those used in the MBSS summer habitat assessment. Currently, SCA data are available for the Ballenger Creek, Breton Bay, Georges Creek, Liberty Reservoir and Upper Patuxent River basins. Additional basins will be added on a regular basis.

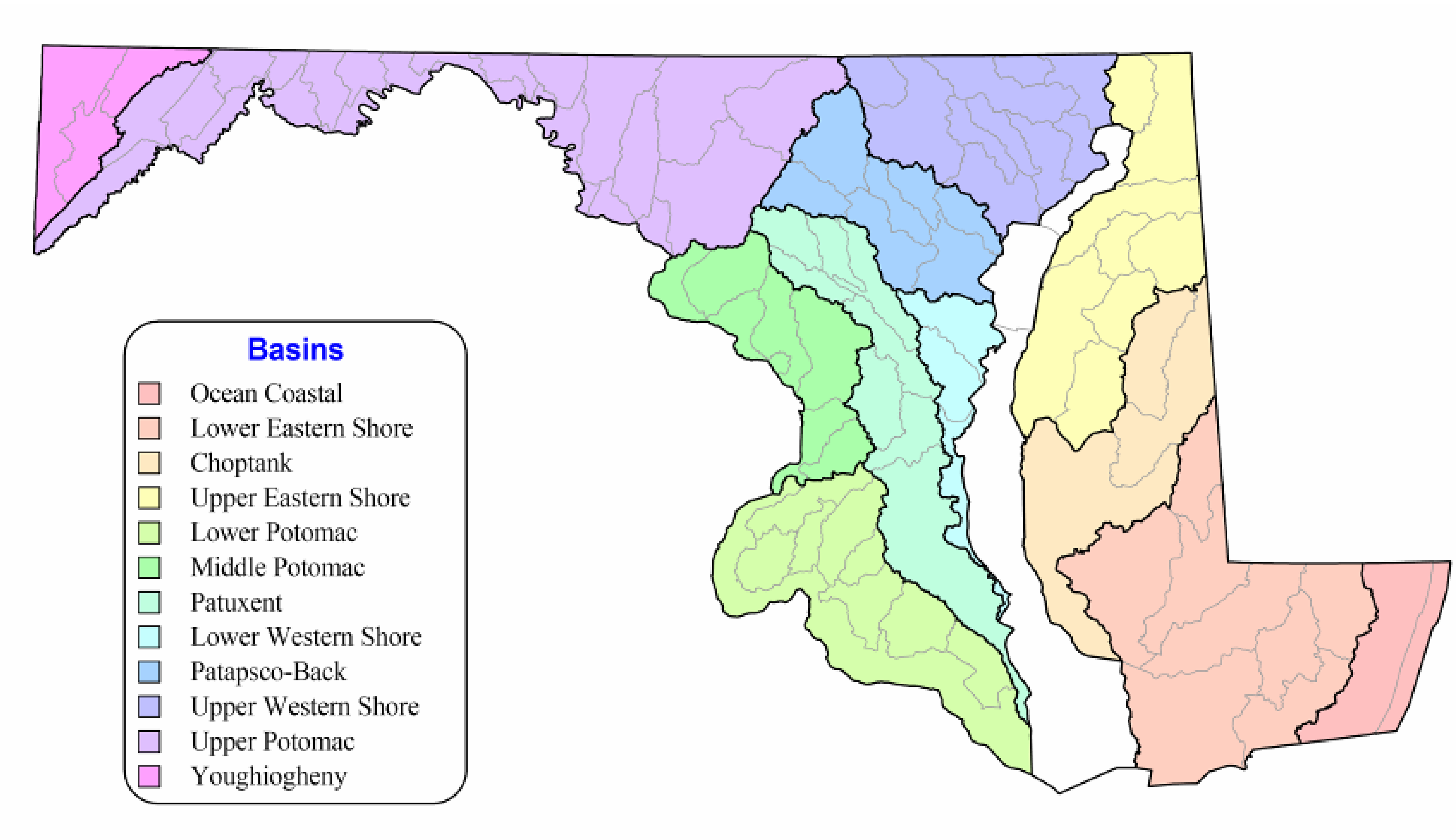


Figure 14-3. Maryland Tributary Strategy Basins with their constituent Maryland 8-digit basins.

resulting from human activities and can help address this issue. More importantly, when biological data themselves can be used, they integrate the cumulative effects of many stressors and are the best evidence of adverse impacts. Investigation into interactions among stressors remains an important area for further research.

The next five sections in this volume address the following important stressor categories:

- Acidification
- Nutrients
- Physical habitat
- Biotic interactions (non-native and invasive aquatic biota)
- Land use

Subsequent sections discuss the relative contributions of each stressor and their cumulative impact on stream resources, and provide basin and site examples of stressor identification.

14.2 Acidification

One of the primary objectives of the MBSS is to assess the effect of acidic deposition on biological resources of Maryland streams–an objective driven by previous studies that document effects on ecologically and economically important species in Maryland's tidal and freshwater ecosystems. Acidification is well known to have detrimental effects on fish assemblages and other aquatic biota (Baker and Christensen 1991, Baker et al. 1990a, 1996) both from the direct effects of low pH and through toxic effects resulting from increases in elemental concentrations (e.g., aluminum, zinc, and mercury) that leach from soils.

In the early 1980s, DNR recognized that atmospheric deposition resulting from the generation of electric power is one of the State's most important environmental problems. The link between acidification of surface waters and acidic deposition resulting from pollutant emissions is well established. To determine the spatial extent of acidification of Maryland streams resulting from acidic deposition, DNR conducted the Maryland Synoptic Stream Chemistry Survey (MSSCS) in 1987 (Knapp et al. 1988). The MSSCS estimated the number of streams affected by, or sensitive to, acidification statewide, and concluded that the greatest concentration of fishery resources at risk may be in streams throughout the Appalachian Plateau and the Southern Coastal Plain physiographic provinces, as well as certain portions of the Blue Ridge.

Because the MBSS collects both biological and water chemistry data, it has the ability to measure not only the spatial extent of acidification in Maryland but also the severity and extent of potential adverse effects on aquatic biological communities.

14.2.1 Background

Acidic deposition occurs as wet deposition (rain, snow, sleet, or hail), dry deposition (particles or vapor), and cloud or fog deposition (common at high elevations and coastal areas). Cloud and fog deposition may significantly contribute to the total deposition of sulfate and nitrogen to high-elevation sites in the northeastern United States (Anderson et al. 1999). Prevailing winds from west to east cause pollutants to be deposited in the mid-Atlantic and Northeast regions. During atmospheric transport, some SO_2 and NO_x will be converted to sulfuric and nitric acids or to ammonium sulfate and ammonium nitrate, all with significant residence times in the atmosphere (Lovett 1994).

The effects of acidic deposition on stream chemistry are well documented (Baker et al., 1990b, 1996, Bricker and Rice 1989, Schindler 1988, Wigington et al. 1996a,b). Conducted in 1987, the MSSCS concluded that approximately one-third of all headwater streams in Maryland are sensitive to acidification or are already acidic (Knapp et al. 1988). Research has demonstrated that the vulnerability of stream systems to acidic deposition depends on basin hydrology and the ability of the vegetation, soils, and bedrock within the basin to buffer acidic inputs.

Defining characteristics of surface waters sensitive to acidification are low to moderate pH and acid neutralizing capacity (ANC), where pH is a measure of the acid balance of a stream. The pH scale ranges from 0 to 14, with pH 7 as neutral. Low to moderate pH (< 6) signifies acidity. ANC is a measure of the capacity of dissolved constituents in the water to react with and neutralize acids, and is used as an index of the sensitivity of surface water to acidification. The higher the ANC, the more acid a system can assimilate before experiencing a decrease in pH. Repeated additions of acidic materials may cause a decrease in ANC. In many acidic deposition studies (e.g., Schindler 1988, Roth et al. 1999), an ANC of 200 µeq/L is considered the threshold for defining acid-sensitive streams and lakes.

In a recent study of acid deposition impacts in Maryland streams (Janicki et al. 1991, Sverdrup et al. 1992), the sensitivity of an indicator species was expressed as the critical pH at which half or more of the population experiences acute or chronic effects (Janicki et al. 1991, Sverdrup et al. 1992, Morgan 1995). The level of acid deposition that results in the critical pH is known as the "critical load". In the critical loads study, information on soil buffering ability was combined with MSSCS ANC values to estimate critical loads at specific sites across the state. Critical load results revealed wide differences in the sensitivity of Maryland streams in different provinces:

- The Appalachian Plateau, Coastal Plain, and portions of the Blue Ridge are very sensitive with critical load values < 0.5 keq SO_4/ha/year (24 kg SO_4/ha/year).

- In contrast, the Valley and Ridge, Piedmont, and portions of the Blue Ridge exhibit critical loads well over 2.0 keq SO_4/ha/year (96 kg SO_4/ha/year). In these regions, limestone bedrock and derived soils are prevalent.

These critical load values provided the basis for a reassessment of acidic deposition in 1998 (Miller et al. 1998). When measured sulfate deposition was compared with critical loads, the results suggested that streams continue to be affected in some areas of Maryland despite recent reductions in industrial sulfate emissions. This was a finding consistent with stream chemistry measured in the 1995-1997 MBSS.

Acidification is known to cause declines in both diversity and abundance of fish populations. Current evidence indicates that the number of aquatic taxa in an ecosystem usually declines with increasing acidity (Eilers et al. 1984, Mills and Schindler 1986, Stephenson and Mackie 1986). In a review of pH effects on aquatic biota, Baker and Christensen (1991) report a number of critical thresholds at which certain fish populations are affected. Many streams in Maryland have pH values below critical levels, with critical pH values for inland species ranging from 5.0 to 6.5 (Baker et al. 1990a; Morgan et al. 1991, Pinder and Morgan 1995). For instance, several bass and trout species have a reported critical threshold of pH 5.0-5.5, while a number of more sensitive cyprinid and darter species are adversely affected at pH 5.5-6.0. Acid-tolerant species, such as the yellow perch *(Perca flavescens),* can survive at pH levels of 4.5 or lower. Eastern mudminnow (*Umbra pygmaea*) have been found in waters with a pH 4.0 or lower (Jenkins and Burkhead 1993), although the acidity may be partially derived from weak organic acids.

The primary mechanisms for fish population declines under acidic conditions include both recruitment failure (owing to an increased mortality of early life stages) and direct effects on adult survival. One of the physiological effects observed when pH decreases is the disruption of the normal internal ionic salt balance, which causes the fish to lose salt to the surrounding water. If the salt losses exceed intake, fish go into shock, lose equilibrium and eventually die from circulatory collapse-an osmoregulatory process. Acidic waters may also inhibit development of fish reproductive organs and facilitate development of a mucous that suffocates eggs and fry (Eno and Di Silvestro 1985). The loss of entire fish populations in abnormally acidic streams or lakes usually occurs because of successive failures in the reproductive cycle (Baker 1996, Carline et al. 1992, 1994). Other detrimental effects are caused by the increased concentrations of metal ions resulting from acidification (e.g., from the leaching of aluminum and the formation of methylmercury).

In addition to potential long-term (chronic) acidification, streams in Maryland are susceptible to rapid, short-term increases in acidity (episodic acidification) related to precipitation, snowmelt, and stormflow events (Greening et al. 1989, Gerritsen et al. 1992, Wigington et al. 1993, Eshleman et al. 2000). One study estimates that 50% more streams in the northern Appalachian Plateau of Western Maryland probably experience the deleterious effects of episodic acidification than are chronically acidified (Eshleman 1995). Spatial and temporal variability of acidic conditions are important to the magnitude of effects on aquatic biota. For example, a pulse of episodic acidification during juvenile recruitment could have a greater effect on a fish population than it would at other times of the year. The highest levels of acidity in Maryland streams have been recorded in the spring, when many fish, including economically important anadromous fish species of the Chesapeake Bay, enter the freshwater portions of coastal streams to spawn. Large-scale fish kills frequently result when snowmelts and large quantities of acidic materials are released into rivers and streams (Eno and Di Silvestro 1985, Molot et al. 1989, Baker et al. 1996, Carline et al. 1992).

Because many invertebrate taxa are also sensitive to acidification, detrimental effects on food webs may occur well before direct toxicity to fish is evident (Schindler et al. 1989, Gill 1993). Benthic invertebrate taxa richness may be reduced as a result of acidification (Ford 1988), but this loss may be compensated for by an increase in numbers of acid-tolerant species resulting in little or no decrease in overall biomass (Eriksson et al. 1980, Dixit and Smol 1989). Some invertebrate taxa-notably mollusks, crustaceans, leeches, mayflies, some species of water striders, caddisflies, damselflies, dragonflies, and cladocerans-are sensitive to acidification and become scarce or disappear between pH 5.0 and 6.0 (Havas and Hutchinson 1982, Eilers et al. 1984, Raddum and Fjelheim 1984, Ormerod and Tyler 1986, Bendell 1988, Bendell and McNicol 1987).

14.2.2 Extent of the Acidification Problem

Both rounds of the MBSS measured several water quality parameters related to acidification during both the spring baseflow index and summer baseflow index periods (see Volume 6: Laboratory, Field, and Analytical Methods). Thresholds for pH and ANC were defined using U.S. NAPAP (1991) conventions and statistical distributions of each water quality parameter (Table 14-1). The high pH

Table 14-1. MBSS water quality thresholds for pH and ANC as measured in the first and second rounds of the MBSS.

Parameter		Low	Moderate	High
pH		< 5.5	5.5 - 6.5	> 6.5
ANC	**Acidic**	**Chronic**	**Episodic**	**Normal**
μeq/L	< 0	0 – 50	50 - 200	> 200

and normal ANC thresholds were broken down further into very high for a pH greater than 7.5, and high for an ANC greater than 750 μeq/L. This was done for acidification analyses by the ten Maryland tributary strategy basins (plus the Youghiogheny and Ocean Coastal basins) and for testing biotic relationships.

14.2.2.1 Low pH

In the MBSS, pH was measured in the spring index period as closed pH (measured in the laboratory), and during the summer index period using field meters. There is a strong correlation (r^2 = 0.60) between spring and summer pH (Figure 14-4). However, the relationship is not as robust at the lower pH levels (< 6.0), where field pH and closed pH diverge. Closed pH predicts a lower spring index pH for the MBSS sites than in the summer index period sampling. At pH above 6.5, the regression fit is very good, although there is some scatter above pH 8.0.

<u>2000-2004 MBSS Spatial Extent</u> - In Round 2 of the MBSS, five basins – the Lower Eastern Shore, Lower Western Shore, Lower Potomac, Youghiogheny, and Ocean Coastal–had greater than 10% of stream km below a pH of 5.5, the low pH threshold (Table 14-2). However, five basins exceeded 40% of stream km in the pH 5.5–6.5 category, and five basins had greater than 85% of their stream km in the pH greater than 6.5 category, reflecting both their geology and land use. The difference in mean pH among basins is shown in Figure 14-5. Three basins had a mean pH less than 6.5 – Lower Potomac, Lower Eastern Shore, and Choptank. The Ocean Coastal basin

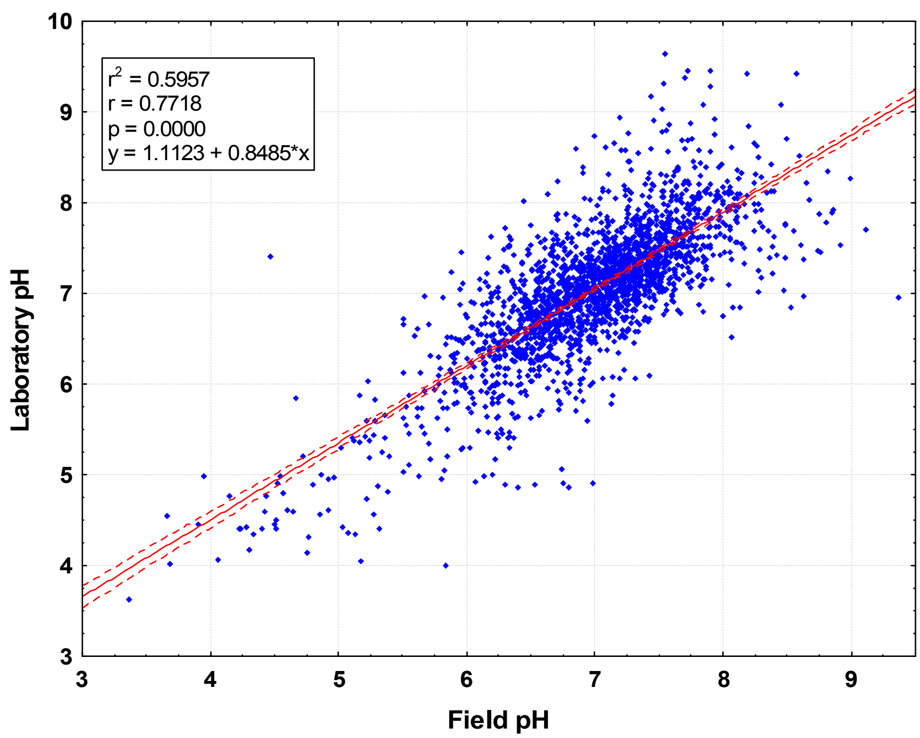


Figure 14-4. Relationship of field pH (summer index period) with laboratory pH (closed pH taken during spring index period).

Table 14-2. Percentage of stream km within each threshold category as defined in Table 14-1 for pH (standard units) and ANC (μeq/L) for all tributary strategy basins. The Youghiogheny and Ocean Coastal basins are included but their drainages are not part of the Chesapeake Bay basin.

		Basin											
Analyte	**Threshold**	**Lower Eastern Shore**	**Choptank**	**Upper Eastern Shore**	**Upper Western Shore**	**Patapsco/ Back River**	**Lower Western Shore**	**Patuxent**	**Lower Potomac**	**Middle Potomac**	**Upper Potomac**	**Youghio-gheny**	**Ocean Coastal**
pH	< 5.5	26.2	4.0	1.3	0.8	1.6	10.0	2.7	14.4	0.5	3.4	16.1	11.1
	5.5 – 6.5	40.0	43.9	20.9	4.2	1.6	43.5	12.3	42.2	3.9	4.1	11.9	55.6
	> 6.5	33.7	52.1	77.8	95.0	96.9	46.5	85.0	43.4	95.6	92.4	72.0	33.3
ANC	< 0	12.2	2.7	1.3	0	0	3.0	2.7	4.3	0.5	2.1	10.2	11.1
	0 – 50	11.8	1.3	0	0.8	1.6	7.0	2.2	23.2	0	3.9	9.2	0
	50 - 200	37.5	40.4	16.6	8.1	3.4	39.9	13.8	47.7	6.6	15.5	59.5	11.1
	> 200	38.6	55.7	82.1	91.1	95.0	50.1	81.3	24.7	92.9	78.5	21.1	77.8

consists of only eight samples, collected in the second round in the large pH sampling effect. This reflects the difficulty in site selection due to the tidal influence of the Maryland coastal bays and the small basin.

The spatial extent of pH in MBSS streams is depicted in Figure 14-6. Four clusters of low and moderate pH values stand out – the Eastern Shore, the southern western Coastal Plain, portions of the Blue Ridge associated with the Catoctin Mountains, and true western Maryland, in both the Youghiogheny and western North Branch drainage. For the latter two basins, acid mine drainage (AMD) is a confounding factor in acidification assessment.

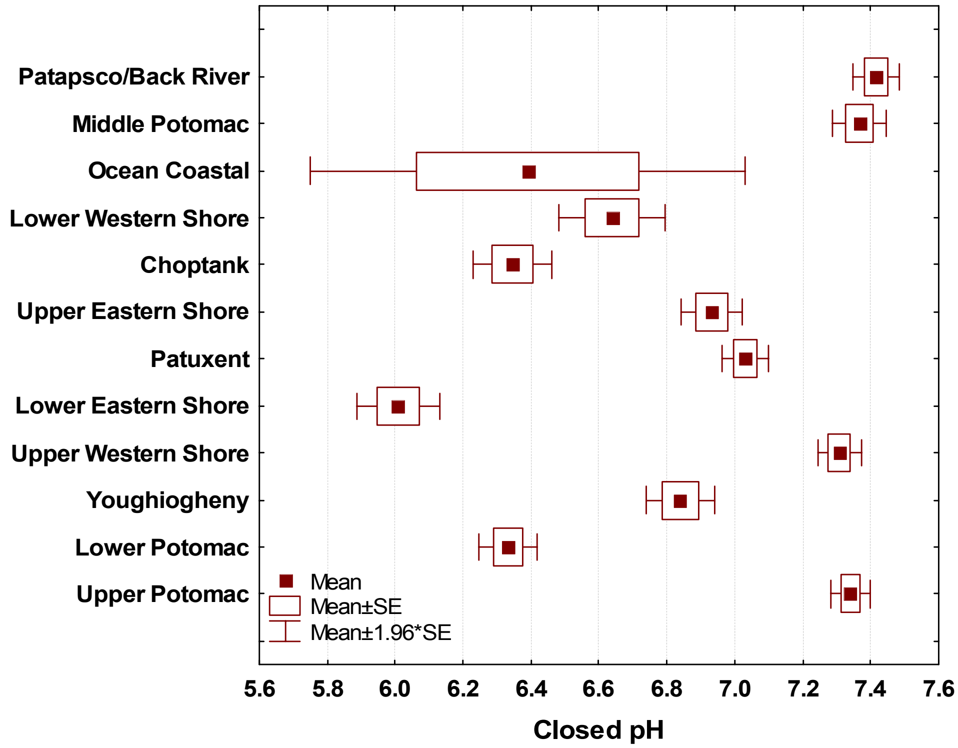


Figure 14-5. Closed pH for all Tributary Strategy Basins (includes Youghiogheny and Ocean Coastal) of Maryland.

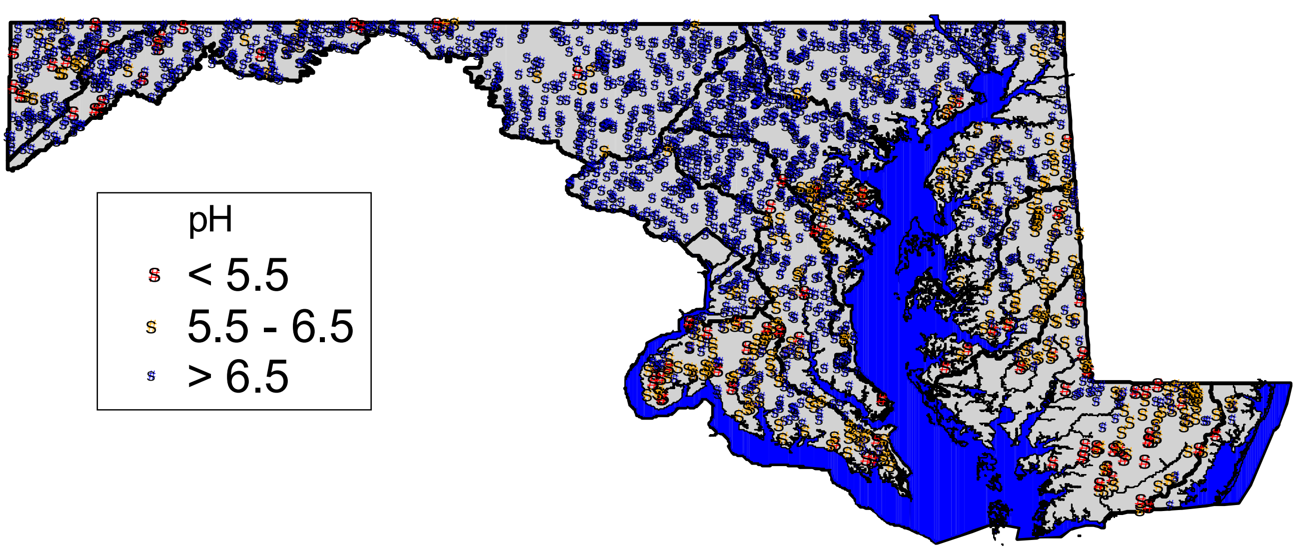


Figure 14-6. Spatial distribution of MBSS sites for pH values taken during the MBSS spring index period.

1995-1997 MBSS Spatial Extent - During spring sampling, an estimated 2.6% of the stream km sampled in the 1995-1997 MBSS had a pH less than 5, while another 6.4% had a pH of 5-6. Low spring pH was most common in the Pocomoke, where about 34% of stream km had a pH less than 5 and 28% of stream km had a pH of 5-6. Summer field sampling results were similar. An estimated 1.8% of the stream km had a pH less than 5, while 4.1% had a pH of 5-6. The lowest summer pH was in the North Branch Potomac, where about 16% of the stream miles had summer pH less than 5 and 1% had summer pH 5-6. Small streams appear to be most susceptible to low pH conditions; first-order streams have the highest percentage of stream km in the low pH classes. None of the third-order sites sampled had spring pH < 5. During spring, only 2.7% of third-order stream km had a pH of 5-6, compared to 8.4% of first-order stream km. Likewise, only 1.6% of third-order stream km sampled in summer had pH <6 as compared to 7.3% of first-order stream km.

14.2.2.2 Low Acid Neutralizing Capacity (ANC)

In the MBSS, ANC is measured in the spring index period. The definition of ANC, from Stumm and Morgan (1996), is:

ANC is the concentration of proton acceptors in solution minus the concentration of proton donors

or,

$$ANC = [HCO_3^-] + 2[CO_3^{2-}] + [OH^-] + [\text{other H+ acceptors}] - [\text{H+ donors}] \quad (1)$$

where, HCO_3^- is the bicarbonate ion, CO_3^{2-} is the carbonate ion, and OH^- is the hydroxide ion - all measured in molar concentrations. An alterative equation, simply the summation of all base cation concentrations minus the summation of all strong acid anion concentrations, is:

$$ANC = \Sigma C_B - \Sigma C_A$$

For most lakes and streams, all terms except bicarbonate are insignificant. In low ANC waters, the concentration of other hydrogen acceptors becomes important, including organic substances and aluminum species. Important noncarbonate contributors may include organic ligands (especially acetate and propionate) as well as hydroxide, silicate, borate, and less commonly ammonia and sulfide (Hem 1985); phosphate and arsenate may contribute to ANC as well (Stumm and Morgan 1996).

ANC is customarily determined for an unfiltered water sample composed of solute plus particulars. ANC is usually reported as either meq/L or µeq/L. An important point regarding the ANC unit is that it may be measured below zero since this analysis is done as a Gran titration. ANC is an important measure of acidification because it may be used as an index to estimate which aquatic systems may become acidified under either chronic or episodic conditions. Aquatic systems with an ANC < 0 µeq/L are acidic, 0 ≤ ANC < 50 µeq/L are highly sensitive to acidification (chronic), 50 ≤ ANC < 200 µeq/L are sensitive to acidification (episodic), and ANC > 200 µeq/L are not sensitive to acidification (Table 14-1).

There is a strong curvilinear relationship of pH versus ANC (Wigington et al. 1990, 1993, U.S. National Acid Precipitation Assessment Program – NAPAP 1991). This common ANC pattern is present for both MBSS rounds (Figure 14-7). Above an ANC of 200 µeq/L, pH is generally greater than 6.5, and slowly increases with rising ANC up to about pH 8.3 (bicarbonate saturation) for aquatic systems (Allan 1995). Below an ANC of 100, pH drops rapidly and falls to about pH 5.0 at 0 µeq/L, and continues to drop below 0 µeq/L. At low ANC levels, dissolved organic carbon (DOC) may be an important acid contributor, having a contribution of weak organic acids (Figure 14-7). DOC concentrations greater than 5.0 mg/l may contribute to low ANC levels, and high DOC levels (> 10 mg/l) obviously contribute.

2000-2004 MBSS Spatial Extent – Three basins–the Lower Eastern Shore, Youghiogheny and Ocean Coastal–had greater than 10% of stream km with an ANC < 0 µeq/L, indicating acidic conditions (Table 14-2). In addition, the Lower Eastern Shore, Lower Potomac and Youghiogheny also had significant stream km in the 0-50 µeq/L ANC range. The Upper Western Shore, Patapsco/Back River, and Middle Potomac had greater than 90% of stream km with an ANC greater than 200 µeq/L. Six basins had a mean ANC greater than 500 µeq/L and another three basins had a mean ANC between 200-400 µeq/L (Figure 14-8). The Lower Potomac, Lower Eastern Shore, and Youghiogheny basins had a mean ANC less than 200 µeq/L, with a significant number of stream km with ANC < 0 µeq/L in the Lower Eastern Shore and Youghiogheny.

The spatial extent of low ANC MBSS streams is depicted in Figure 14-9. Four clusters of acidic and low ANC values stand out: the lower Eastern Shore, the southern Coastal Plain, portions of the Blue Ridge associated with the Catoctin Mountains, and true western Maryland, in both the Youghiogheny and western North Branch drainage.

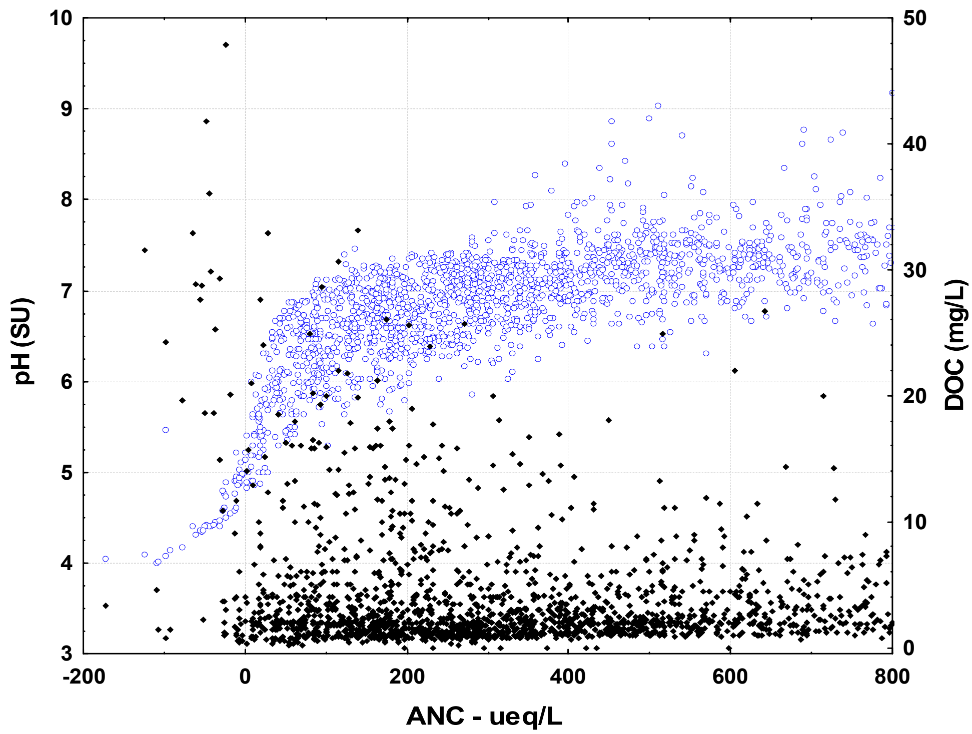


Figure 14-7. Relationship of pH (blue dots) and DOC (black diamonds) with ANC for all MBSS sites.

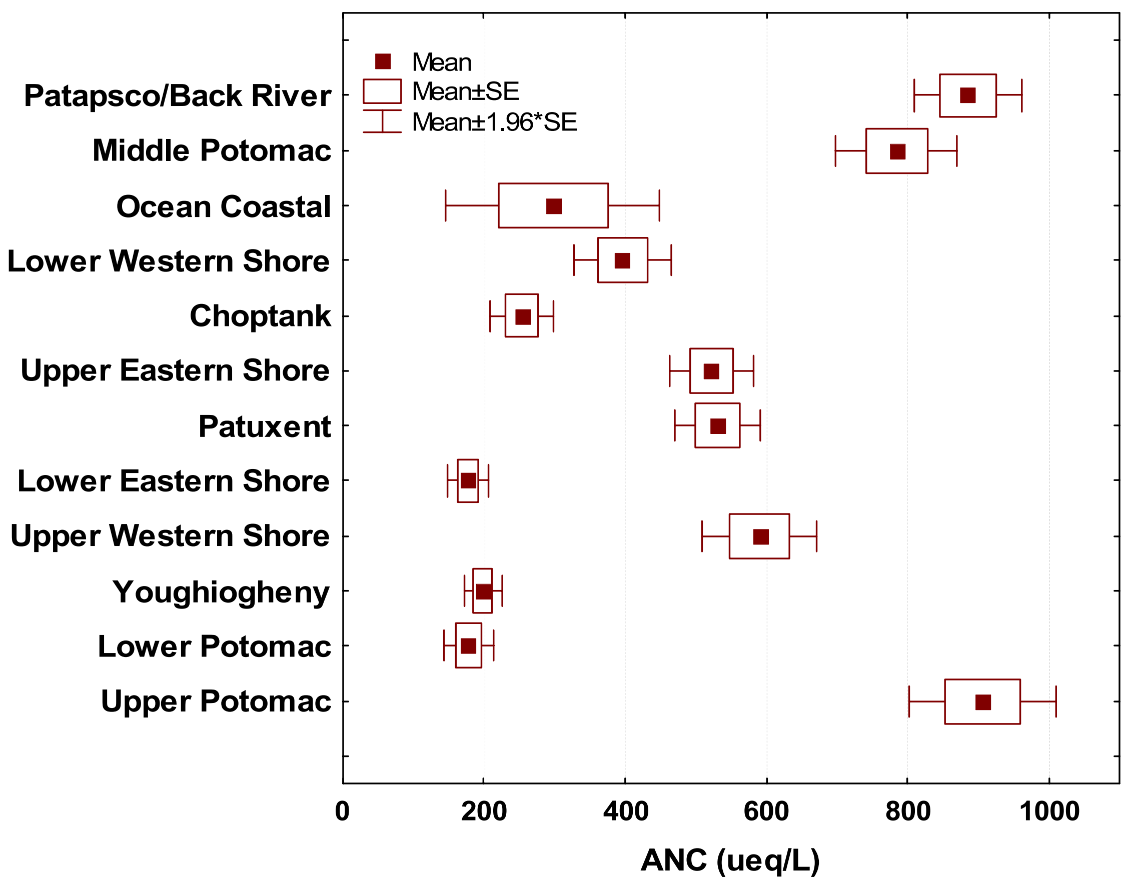


Figure 14-8. ANC (μeq/L) for all Tributary Strategy Basins (includes Youghiogheny and Ocean Coastal) of Maryland.

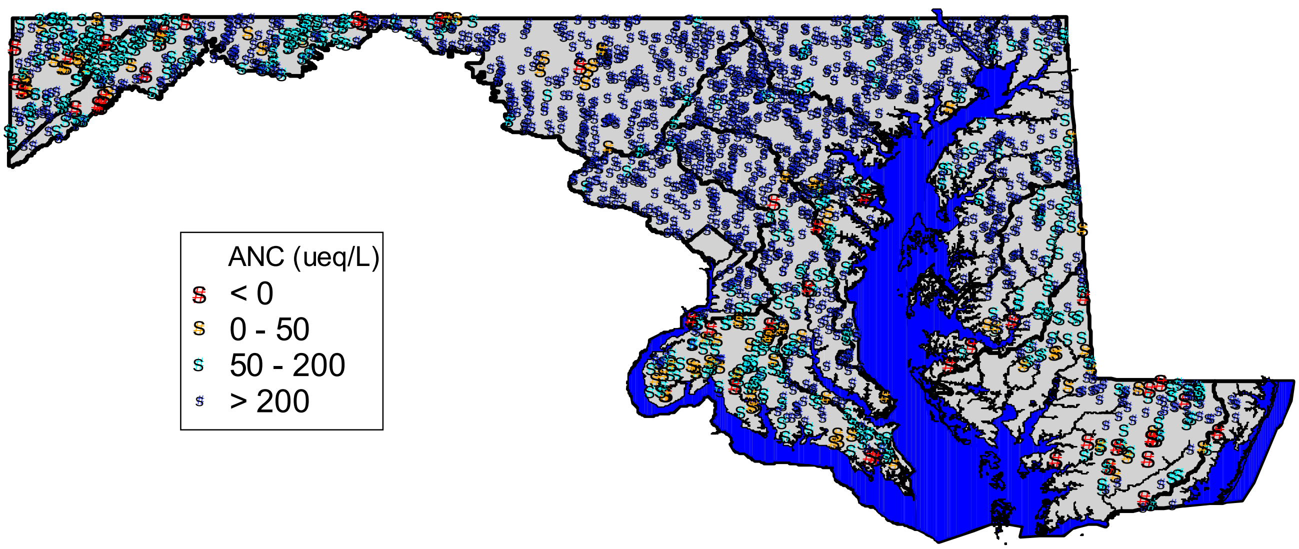


Figure 14-9. Spatial distribution of MBSS sites for ANC taken during the MBSS spring index period.

1995-1997 MBSS Spatial Extent - Statewide, an estimated 28% of the stream km were acidic or acid-sensitive, including about 2% acidic, 4% highly sensitive, and 22% sensitive to acidification in the first round of the MBSS. Statewide, the estimated percentage of stream km with ANC < 0 is 3% of first-order, 2% of second-order, and 0% of third-order stream km. The estimated percentage of stream km with ANC < 200 is 31% of first-order, 21% of second-order, and 20% of third-order stream km.

14.2.3 Sources of Acidity

Acidic deposition, acid mine drainage, agricultural runoff, and natural organic materials all contribute to the observed acidification of Maryland streams. Acidic deposition is the contribution of material from atmospheric sources, both as precipitation (wet) and particulate (dry) deposition. Acidic deposition is generally associated with elevated concentrations of sulfate and nitrate in precipitation (Wigington et al. 1990, 1993, 1996a,b). Acid mine drainage (AMD) results from pyrite oxidation - from mine spoils and abandoned mine shafts - and is known to cause extreme acidification of surface waters, as well as to affect stream physical substrate. Streams strongly affected by AMD exhibit high levels of sulfate, manganese, iron, aluminum, and conductivity. A third source of acidification is surface runoff from agricultural lands fertilized with high levels of nitrogen, or other acidifying compounds. Lastly, the natural decay of organic materials may contribute acidity in the form of organic anions, as in blackwater streams associated with bald cypress wetlands or boreal bogs. Streams dominated by organic sources of acidity are often characterized by high concentrations of dissolved organic carbon (DOC > 8 mg/l) and organic anions.

Acidification sources in Maryland streams have been examined in previous DNR studies using water chemistry data from the MSSCS and other regional surveys. In a study of Maryland Coastal Plain streams, Janicki (1991) reported a predominance of low ANC conditions, and found that differences in stream chemistry within the region were related to land use. In particular, ANC tends to be higher in basins dominated by agriculture. Agricultural activities in Coastal Plain basins may have different effects on stream chemistry, adding both ANC, from soil liming practices, and strong acid anions from nitrogen fertilizers (Janicki et al. 1995). Janicki and Wilson (1994) estimate that acidic deposition is the dominant source of acidity in about 45% of the low ANC streams in the Maryland Coastal Plain, while combined inputs from acidic deposition and agricultural sources affect about 55% of the streams. In Maryland's Appalachian Plateau and Blue Ridge regions, where there are also a significant number of acidic and acid-sensitive streams, bedrock geology is an important factor in determining stream response to acidic deposition, according to analyses by Janicki (1995). Atmospheric deposition is identified as the major source of acidification in the Appalachian Plateau and Blue Ridge streams. Organic acids and agricultural sources did not appear to be major contributors to acidification in western Maryland streams. The analyses by Janicki (1995) did not include any AMD effects.

For the MBSS, a new analysis was conducted to estimate the extent of impacts by acidic deposition, acid mine drainage, agricultural runoff and organic sources. Water chemistry data from sites with low ANC (< 200 µeq/L) were examined to identify dominant sources of acidi-ication and to estimate the percentage of stream km impacted by each according to the flow chart in Figure 14-10. This data set was compared by basins because different acidity sources were expected to be important in the eastern and western parts of Maryland.

Instream concentrations of sulfate and nitrate ions are important indicators of acid sources. For areas near the ocean, however, analyses of stream chemistry need to account for sulfate contributions from airborne sea salts. In this analysis, measured instream sulfate concentrations are corrected for sea salt influence, which decreases with distance from the coast. The amount of marine sulfate is related to levels of marine chloride, which can be estimated from a site's distance from the coast. Because the MBSS does not directly measure chloride concen-trations, estimates of sea salt sulfate and chloride concentrations are made using the following relationships derived for mid-Atlantic streams by the National Stream Survey (Baker et al. 1990b, Kaufmann et al. 1992):

$$\ln(Cl^{-}_{sea}) = 5.4328 - 0.0180*Dist + 0.00004*Dist^{2}$$

$$\text{sea salt corrected } SO_4 = SO_4^{2-}\text{(observed)} - 0.013*Cl^{-}_{sea}$$

where Cl^{-}_{sea}= concentration of sea salt derived chloride (µeq/L), Dist = distance from the coast (km), and SO_4^{2-} (observed) = observed sulfate concentration (µeq/L). The sea salt correction is made only for MBSS sites within 200 km of the ocean. Beyond 200 km, streams are assumed to have no sea salt contributions (Baker et al. 1990b).

In Western Maryland streams, sulfate concentrations are used to distinguish MBSS sites having AMD as the dominant source of acidification from those dominated by acidic deposition (Figure 14-10). Based on results of previous studies in the Mid-Atlantic Highlands streams (Kaufmann et al. 1992, Herlihy et al. 1990), thresholds were established to distinguish which sites were affected by AMD. For all sites in the Youghiogheny and North Branch Potomac River basins with ANC less than 200 µeq/L, those with sulfate concentrations greater than 500 µeq/L are designated as dominated by AMD. Sites with sulfate in the 300-500 µeq/L (~ 14 to 24 mg/l) range are considered affected by both AMD and acidic deposition (Figure 14-10).

Assessment of acidity sources is critical to understanding stream chemistry, especially in regions affected by both acid mine drainage (AMD) and acidic deposition (Kaufmann et al. 1991, Herlihy et al. 1991, Roth et al. 1999). Synoptic water quality investigations are extremely important in describing a population of streams that may be influenced by either acidic deposition or AMD. However, these types of surveys may not fully address the causal mechanisms of acidification (Herlihy et al. 1991) requiring a more extensive set of water quality measurements.

The categories of acid sources described in Figure 14-10 were used to estimate the extent of each source affecting Maryland streams. An estimated 27% (SE = 1.2%) of Maryland stream km had an ANC < 200 µeq/L. State-wide, acidic deposition is by far the most common source of acidifying compounds, being the dominant source at about 21.2% of stream km. AMD is the dominant source at only 1.4% of stream km (only found in the Youghiogheny and Upper Potomac basins), while an additional 3.9% of stream km are affected by agricultural inputs of acidity. Organic sources account for less than 1% of the stream km in Maryland.

As expected, acid sources vary considerably among basins (Table 14-3). In the Lower Potomac and Youghiogheny basins, for example, acidic deposition is the only major source of acidity, accounting for 74.4% and 65%, respectively, of stream km with ANC < 200 µeq/L. The Lower Western Shore and the Lower Eastern Shore also had significant stream km affected by acidic deposition (Table 14-3).

Acid mine drainage is only present in the Youghiogheny (10.9%) and Upper Potomac (2.9%) basins (Table 14-3). AMD is not a problem throughout the Upper Potomac, but is restricted to just the Georges Creek basin and the upper North Branch of the Potomac River (above Westernport).

Statewide, less than 1% of all stream km were dominated by organic sources. These sources were present in the Lower Eastern Shore (7%), Choptank (4%) and Ocean Coastal (11%) basins (Table 14-3) and three other basins with 1% or less. The small number of organically dominated streams led to large standard errors (SE > 100%) in estimating the number of stream km that were organically influenced. Twenty-five sites (76%) had a DOC > 8 mg/l, a level commonly used to characterize blackwater streams (streams rich in organic material and typically acid due to the presence of weakly-dissociated fulvic and humic acids).

Across Maryland, 3.9% of acid affected stream km are classified as agriculturally influenced. Agricultural influences on acidity are most extensive in the Lower Eastern Shore, Upper Eastern Shore and Choptank basins (Table 14-3). Smaller percentages (1.0 to 4.2%) are observed in the Upper Western Shore, Patapsco/Back River, Middle Potomac, Upper Potomac, and Ocean Coastal basins. Agriculture is rarely responsible for extreme acidification: only two agriculturally influenced sites had an ANC < 50 µeq/L, the rest had values of 51-200 µeq/L.

Water Chemistry Measurement (Other Parameters)		Source or Type of Acidification
ANC < 200 μeq/L	No →	None
Yes ↓		
Agriculture > 50% of Catchment Area and NO_3-N > 100 μeq/L (1.4 mg/l NO_3-N)	Yes →	Possible Agricultural Influence (AG)
No ↓		
SO_4 ≥ 500 μeq/L (~24 mg/l) and BASIN = North Branch Potomac or Youghiogheny[1]	Yes →	Dominated by Acid Mine Drainage (AMD)
No ↓		
SO_4 ≥ 300 μeq/L (~14 mg/l) and BASIN = North Branch Potomac or Youghiogheny[1]	Yes →	Mixed – Affected by both AMD and Acidic Deposition (ACID DEP)
No ↓		
Organic Ions > NO_3 + SO_4	Yes →	Dominated by organic sources (ORG)
No ↓		
DOC > 8 mg/l	Yes →	Mixed: Affected by both ORG and ACID DEP
No ↓		
Baseflow ANC 50 - 200 μeq/L[2]	Yes →	Stream vulnerable to episodic acidification
No ↓		
Baseflow ANC < 50 μeq/L[2]	Yes →	Chronic acidification – baseflow ANC may be less than 0 μeq/L[2]

Figure 14-10. Diagnosis of stream acidification sources and types for Maryland streams based on MBSS spring index period water chemistry. Note that more than one water chemistry situation (and acidification source or type) may apply.

Table 14-3. Percentage of stream km for each acid source category as defined for all tributary strategy basins. The Youghiogheny and Ocean Coastal are included but their drainages are not part of the Chesapeake Bay basin.

	Basin											
Acid Source	**Lower Eastern Shore**	**Choptank**	**Upper Eastern Shore**	**Upper Western Shore**	**Patapsco/ Back River**	**Lower Western Shore**	**Patuxent**	**Lower Potomac**	**Middle Potomac**	**Upper Potomac**	**Youghio-gheny**	**Ocean Coastal**
Acidic Deposition	38.1	21.0	9.8	4.3	4.0	49.9	15.0	74.4	3.2	17.6	65.0	11.1
Agriculture	16.5	19.1	7.0	4.2	1.0	0	3.7	0	3.9	1.0	3.0	0
Acid Mine Drainage	0	0	0	0	0	0	0	0	0	2.9	10.9	0
Organic	6.9	4.2	1.1	0.4	0	0	0	0.9	0	0	0	11.1
None	38.6	55.7	82.1	91.1	95.0	50.1	81.3	24.7	92.9	78.5	21.1	77.8

The distribution of acidic sources by stream order (for those streams with less than 200 µeq/L) shows some differences in sources (Figure 14-11). Acidic deposition was the major acid source in all stream orders, influencing 63% to 80%. Agricultural acid sources were associated with 14% of first-order streams, 8% of second-order and third-order streams, and 20% of fourth-order streams. AMD affected between 2 to 7 % of first to third-order streams, but these effects are only present in western Maryland (western Allegany and Garrett Counties).

The percentage of stream miles (ANC < 200 µeq/L) that were associated with each acidic source also varied by basin (Figure 14-12). Among low ANC streams, acidic deposition was the dominant source in all basins (and statewide), either singly or in combination with organic acids and AMD. The Choptank, Lower Eastern Shore, Middle Potomac, and Patapsco/Back River all have high levels of agriculturally produced acidic sources in these basins. AMD is present only in the Upper Potomac and Youghiogheny basins. Organic acids are a major source of acidification in the Lower Eastern Shore, Ocean Coastal, Upper Western Shore, Choptank, and Upper Eastern Shore.

14.2.4 Comparison with the 1987 Maryland Synoptic Stream Chemistry Survey

MBSS results from 2000-2004 may be compared with the previous characterization of low ANC in Maryland streams by the 1987 MSSCS (Knapp et al. 1988; Table 14-4). The MSSCS estimated the percentage of stream miles below certain threshold levels of ANC across Maryland, within MSSCS sampling strata derived from physiographic provinces for Maryland. MSSCS measurements were taken in 1987, a dry year that received an average of 11% less rainfall than normal (NOAA 1987). The MSSCS estimated that the greatest concentrations of acidic or acid-sensitive streams were in the Southern Coastal Plain (74% of stream miles) and the Appalachian Plateau (53%), with a statewide estimate of 33.3% of all stream miles below an ANC of 200 µeq/L, 10% below 50 µeq/L, and 3.6% below 0 µeq/L.

There are some important methodological differences between the 1987 MSSCS and the 2000-2004 MBSS. For example, MSSCS sampling was conducted statewide in a single year, while MBSS basins, in the second round, were sampled over a five-year period. Also, the sample frame for the MSSCS specifically excluded streams known to be affected by acid mine drainage, while the MBSS does not exclude these streams. To rectify these differences, the MBSS data were re-stratified by sampling strata, excluding sites that showed AMD as a contributing source of acidity (Table 14-4).

Among basins sampled in the MBSS, ANC patterns are generally consistent with the results of the earlier MSSCS. Sites in the Appalachian Plateau and Valley and Ridge sampling strata had a higher occurrence of acidic or acid-sensitive stream miles, compared to findings from 1987 for these regions. For example, 76% of streams in the Appalachian Plateau category had an ANC below 200 µeq/L in the second round MBSS, versus 53% in the MSSCS. There was an increase in the number of stream miles (30% versus 1.5%) with an ANC less than 200 µeq/L for the Valley and Ridge sampling strata.

These differences may be attributable to the greater number of small streams sampled by the MBSS than MSSCS. These results are consistent with low critical loads estimated for these provinces by Janicki et al. (1995), based on basin hydrology, and the buffering abilities of soils and parent material. Sites in the Piedmont had a low occurrence of low ANC streams in both MSSCS and MBSS sampling; these regions are thought to have higher critical loads values (Janicki et al. 1995). The Blue Ridge province showed a significant difference in ANC results between the MSSCS and MBSS sampling. This difference should be interpreted with caution, because the Blue Ridge is a small region and naturally has large statistical variation in results. The Northern Coastal Plain had similar values between the MSSCS and the second round of the MBSS, while the Southern Coastal Plain showed a drop (74% versus 56%) in streams with ANC less than 200 µeq/L. Across all provinces, the MBSS results show a lower percentage of low ANC sites than do the MSSCS results (from 33% to 27%). This suggests an overall improvement in the acidic condition of Maryland streams from 1987 to 2004. This, however, may be confounded by the design differences described above.

14.2.5 Associations Between Acidification and Biological Condition

Biological data for MBSS sites within designated pH and ANC classes were compared to determine the effect of acidic conditions (primarily acidic deposition) on Maryland stream communities. Acidification of streams may cause declines in the biotic integrity of fish assemblages, as a result of the loss of fish species sensitive to acidification, increases in acid-tolerant species, or the total elimination or reduction in the overall abundance of biota (Baker et al. 1990a, Carline et al. 1992, Van Sickle et al. 1996, Webb et al. 1989, Gagen et al. 1994, Heard et al. 1997).

Streams sensitive to acidification may experience intermittent periods of low pH that may be harmful to fish populations. In particular, streams may be subject to

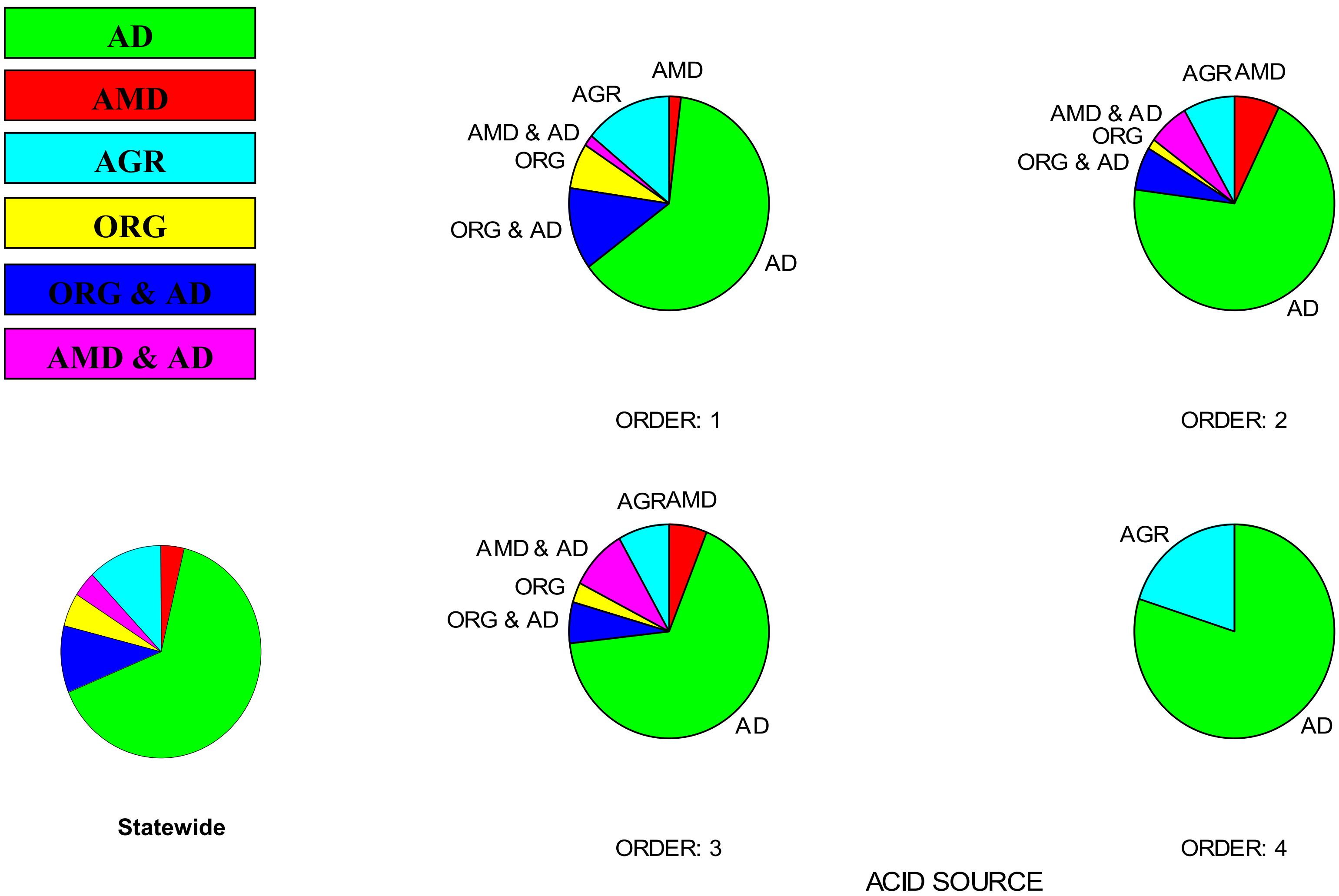


Figure 14-11. Summary of acid sources statewide and by stream order for MBSS sites having an ANC less than 200 μeq/L. (AMD = acid mine drainage; AD = acid deposition; AGR = agriculture; ORG = organic; AMD & AD = mixed sources of acid deposition and acid mine drainage; and ORG & AD = mixed sources of organic and acid deposition).

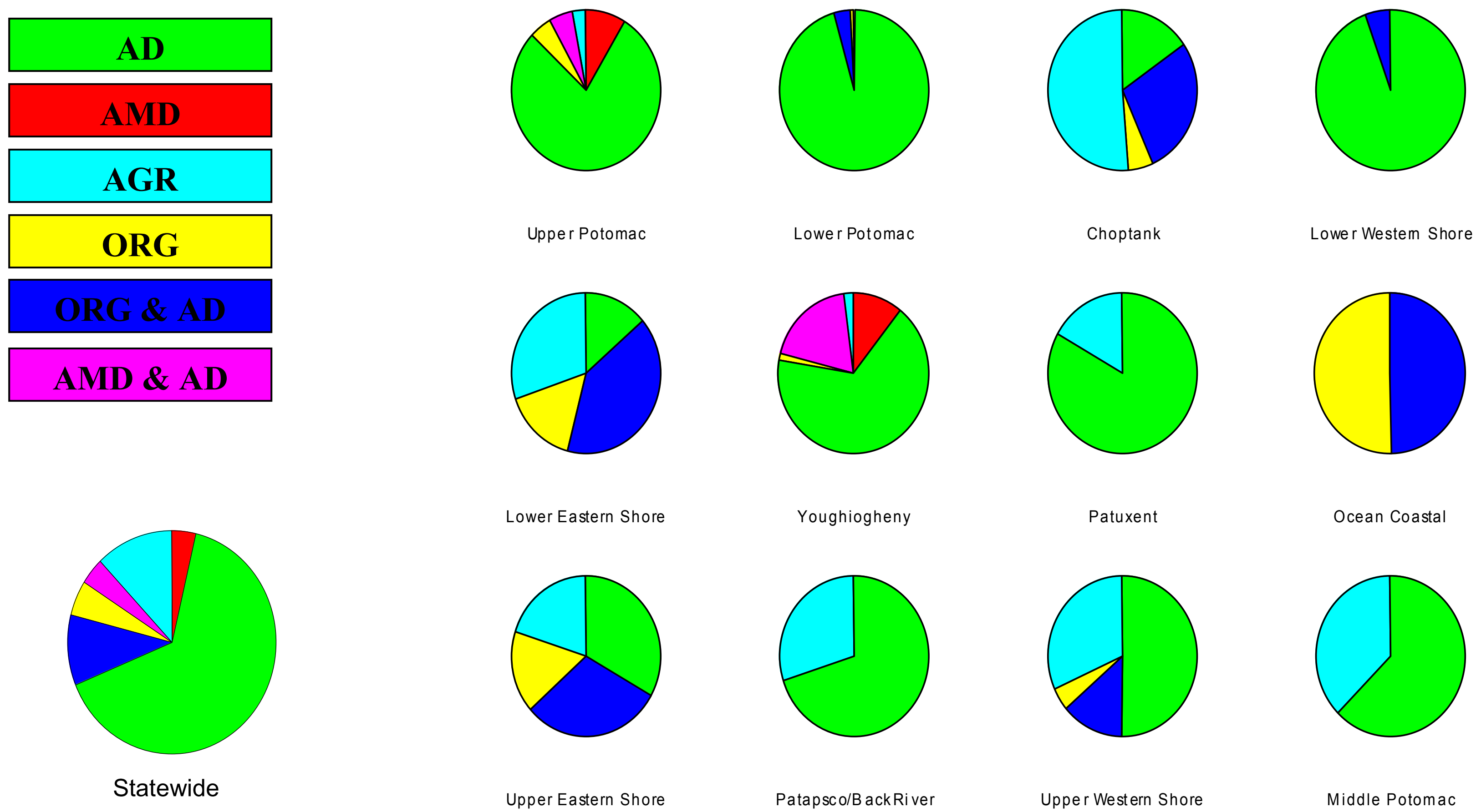


Figure 14-12. Summary of acid sources statewide and by Tributary Strategy Basin (including the Youghiogheny and Ocean Coastal) for MBSS sites having an ANC less than 200 µeq/L. (AMD = acid mine drainage; AD = acid deposition; AGR = agriculture; ORG = organic; AMD & AD = mixed sources of acid deposition and acid mine drainage; and ORG & AD = mixed sources of organic and acid deposition).

Table 14-4. Percentage (and SE) of acidic and acid-sensitive stream miles, as estimated by the 1987 Maryland Synoptic Stream Chemistry Survey (MSSCS) and 2000-2004 MBSS. Estimates are the percentage of stream miles below threshold ANC values, by MSSCS sampling strata.

ANC (µeq/L)	MSSCS SAMPLING STRATA													
	Appalachian Plateau		**Valley and Ridge**		**Blue Ridge**		**Piedmont**		**Northern Coastal Plain**		**Southern Coastal Plain**		**All**	
	N = 139		**N = 47**		**N = 50**		**N = 125**		**N = 99**		**N = 99**		**N = 559**	
	Percent	**SE**	**Percent**	**SE**	**Percent**	**SE**	**Percent**	**SE**	**Percent**	**SE**	**Percent**	**SE**	**Percent**	**SE**
<0	10.7	3.6	0	0	0	0	0	0	2.1	1.5	7.6	2.9	3.6	0.9
<50	15.7	3.9	0	0	5.8	2.5	0.9	1.0	4.7	2.8	29.3	4.7	10.0	1.4
<200	53.3	4.6	1.5	1.3	26.0	5.7	8.9	3.6	28.3	5.2	74.4	5.0	33.4	2.2
	N = 95		**N = 135**		**N = 124**		**N = 438**		**N =**		**N =**		**N = 1,387**	
<0	9.1	3.9	2.2	1.3	2.6	2.6	0.0	0.0	1.8	1.0	6.9	1.5	2.8	0.5
<50	19.5	4.5	4.3	1.7	8.8	3.6	0.2	0.2	2.8	1.1	20.8	2.6	7.5	0.8
<200	75.8	4.8	30.0	3.7	14.2	6.4	5.8	1.6	27.6	3.8	56.4	3.0	27.3	1.2

episodic acidification during springtime, when larval and juvenile fish are particularly vulnerable to adverse changes in water quality (Carline et al. 1992, Gagen et al. 1994, Heard et al. 1997). The MBSS study design did not focus on sampling during high stream flow events that may have produced low pH episodes. Instead, the MBSS results reflect an indirect relationship between episodic acidification and loss of biotic integrity.

Fish IBI and Fish Biomass - The fish index of biotic integrity (FIBI), recently re-developed (Southerland et al. 2005), provides a quantitative (scores of 1 to 5 with < 3 reflecting degradation) biological indicator based on reference conditions. Different fish IBIs were developed for Coastal Plain, Eastern Piedmont, warmwater Highlands, and Coldwater Highlands, and include 4 to 6 metrics each (e.g., number of benthic species). An analysis of FIBI scores shows a significant decline ($p = 0.001$) at both low pH and low ANC classes (Figure 14-13). The low pH (< 5.5) and low ANC class (< 0) have very low FIBI scores, with both mean FIBI scores below 2.0. As pH and ANC increase, by class, the FIBI improves. The highest ANC class (> 750 μeq/L) had a mean FIBI below 3.0.

Fish biomass (g) varied significantly ($p < 0.001$) with both pH and ANC classes (Figure 14-14). For MBSS sites in the low pH and low ANC classes, mean fish biomass was less than 500 g. As pH and ANC increase, fish biomass increases. In the high pH class and the normal and high ANC classes, fish biomass averages over 3000 g/MBSS site. Normally, gamefish, including brook trout, do not persist where ANC is < 0, therefore their biomass drops to zero in that class.

The MBSS results are merely a snapshot of acidity (pH and ANC) and biological condition at one point in time and do not capture episodic acidification contributing to the uncertainty in the relationship among pH-ANC, the FIBI, and fish biomass.

Benthic IBI, Benthic Taxa, and Ephemoptera, Plecoptera, and Trichoptera (EPT) Taxa - Three measures of the benthic macroinvertebrate community, the benthic IBI (BIBI), also re-developed (Southerland et al. 2005), the number of benthic taxa, and the number of EPT taxa were compared among pH and ANC classes. The BIBI combines 6 to 8 metrics separately for Coastal Plain, Eastern Piedmont, and Highlands, but the number of benthic taxa and EPT taxa are themselves robust indicators of stream condition (Resh and Jackson 1993, Stribling et al. 1998).

The response pattern for the BIBI versus pH and ANC shows declines of both high and low values (Figure 14-15). As expected, the mean BIBI is below 3.0 for the low pH and acidic ANC classes, indicating an acidification effect on the benthic community, above 3.0 for the moderate and high pH classes, and above 3.0 for the chronic, episodic and normal ANC classes. However, for the very high pH class (greater than 7.5) and the high ANC class (greater than 750 μeq/L) the BIBI is well below the 3.0 threshold. This BIBI response may indicate species shifts at higher ANC and pH, not captured by BIBI reference conditions, or it may indicate that there are confounding factors operating at these higher pH and ANC levels.

This same relationship of low benthic scores within both low and high values of pH and ANC is duplicated for both the number of benthic taxa and the number of EPT taxa.

An analysis of the new Physical Habitat Index (PHI) (Paul et al. 2003) indicates that the very high pH class (PHI = 63.1) is significantly lower than observed in the low (74.8), moderate (69.2), and high (68.1) pH classes. For PHI, the high ANC class (60.6) is also significantly lower than the acidic (75.9), chronic (76.2), episodic (71.9) and normal ANC (67.3) classes. This suggests that there are confounding factors influencing biotic responses to acidification in the very high pH and high ANC classes.

Based on comparing pH and ANC to the FIBI, biomass, BIBI, number of benthic taxa, and number of EPT taxa, an ANC threshold of 50 μeq/L (pH = 6.16) is adequate for defining acute and chronic effects of acidification (MacAvoy and Bulger 1995). However, other investigators propose an ANC of 20 μeq/L (pH = 5.72) as a critical tolerance level for fish and invertebrates (Lien et al. 1996).

14.2.6 Fish Tolerance to Low pH Conditions

The presence of fish species at low summer pH (pH < 6.5) MBSS sites indicates which species were most tolerant of summer acidic conditions (Table 14-5). Many of these species were previously reported as tolerant to low pH conditions (Graham 1993, Baker and Christensen 1991, Baker et al. 1990), although not all Maryland fish species were covered by these earlier studies. Interestingly, brook trout are not present below a summer pH of 6.0; this species is often considered to be acid tolerant, being found at pH values as low as 5.2 – 5.3 (Baker et al. 1990, 1996; Carline et al. 1992, Webb et al. 1989). In general, the results for rare species should be interpreted with caution as geographic or other factors may be limiting their occurrence in some pH classes.

Of the 66 species listed (*Lepomis* hybrid is not included in percentages), 12% of the species are found at a summer pH < 5.0, 34% in the pH range of 5.0 to 5.5, 65% in the pH 5.5 to 6.0 range, and 98% in the pH range of 6.0 to 6.5. Low pH tolerant species, for summer MBSS collections, include a number of species such as pirate perch, bluespotted sunfish, and mud sunfish that are

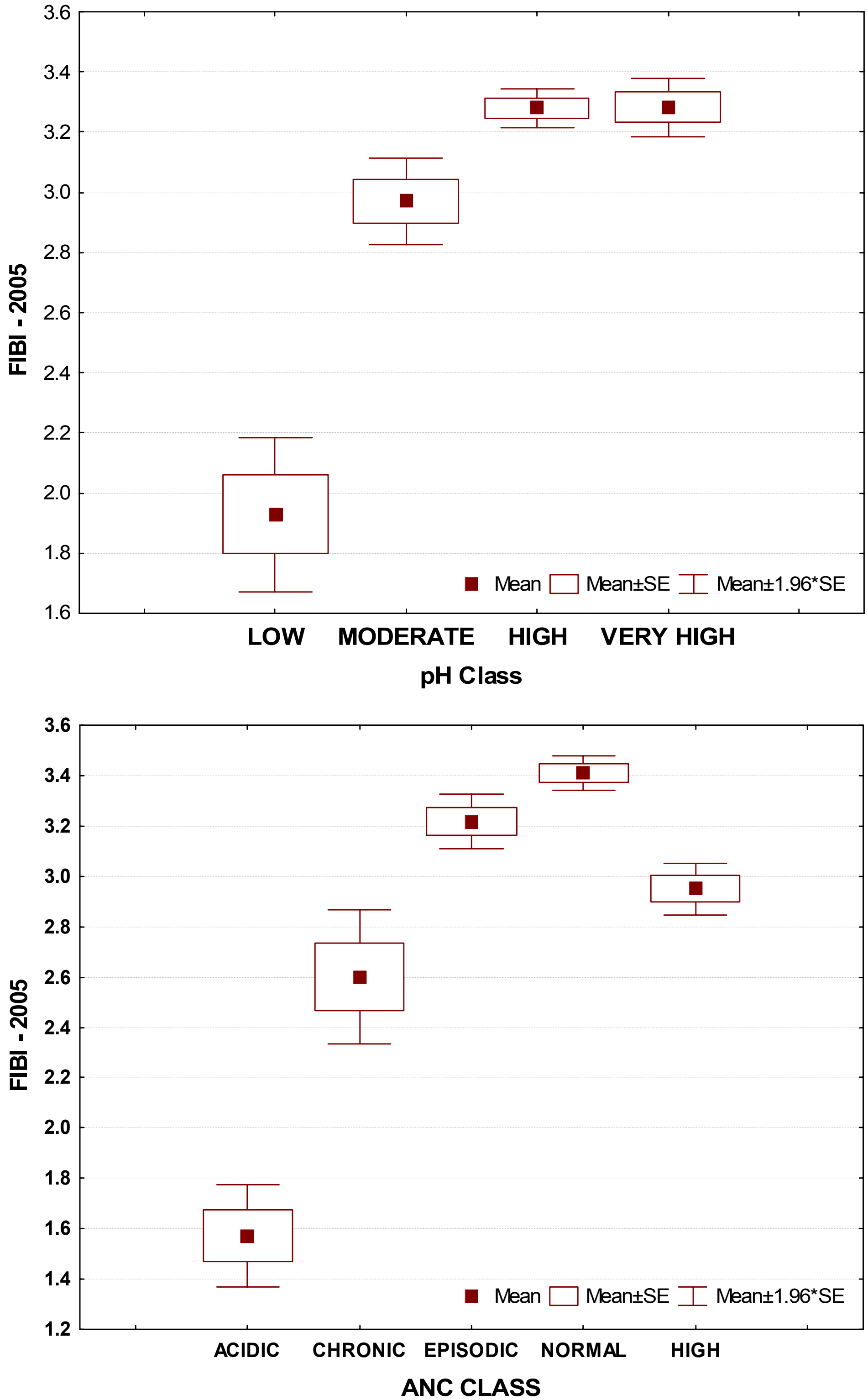
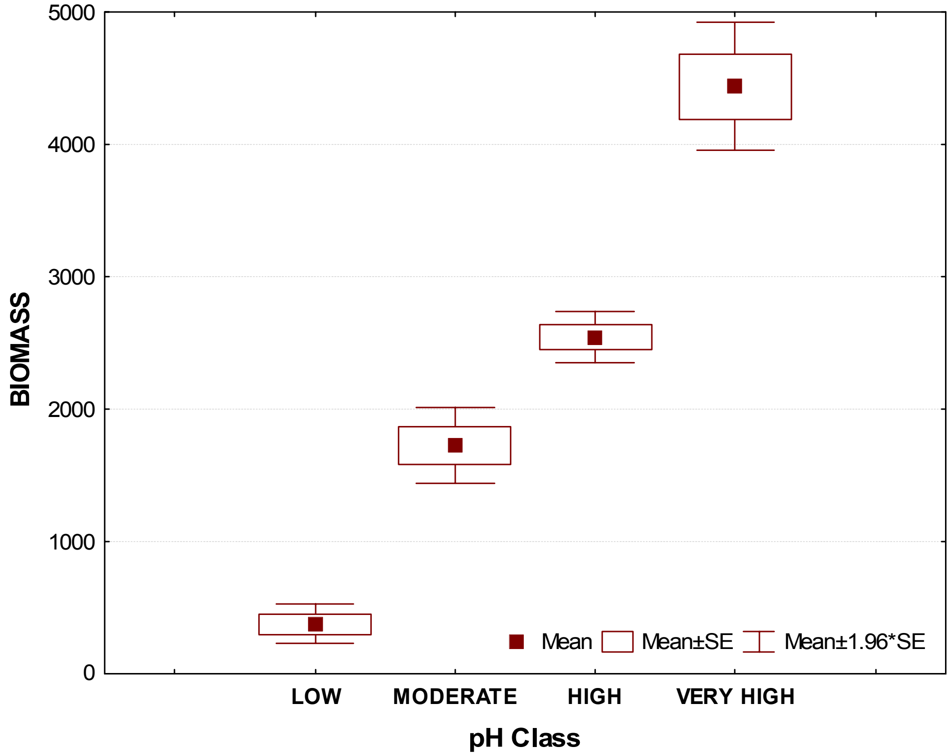


Figure 14-13. Summary of FIBI (2005) for four pH classes (upper) and five ANC classes (below) for MBSS sites. Classes are described in Table 14-1 with the addition of very high pH > 7.5 and high ANC > 750 µeq/L classes.



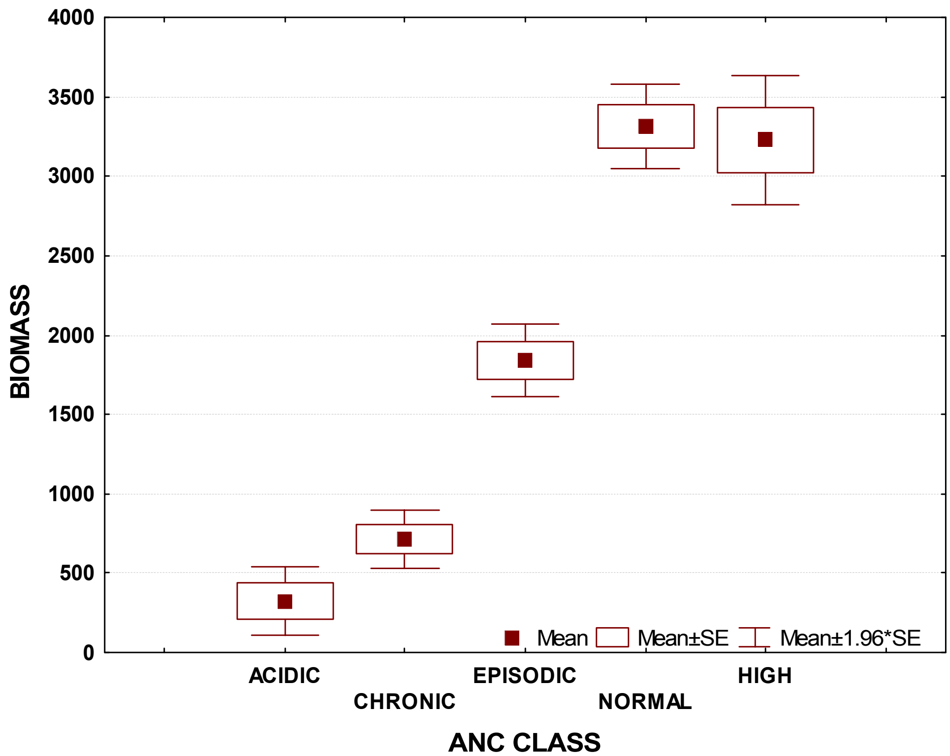


Figure 14-14. Summary of fish biomass (g) for four pH classes (upper) and five ANC classes (below) for MBSS sites. Classes are as described in Table 14-1, with the addition of very high pH > 7.5 and high ANC > 750 µeq/L classes.

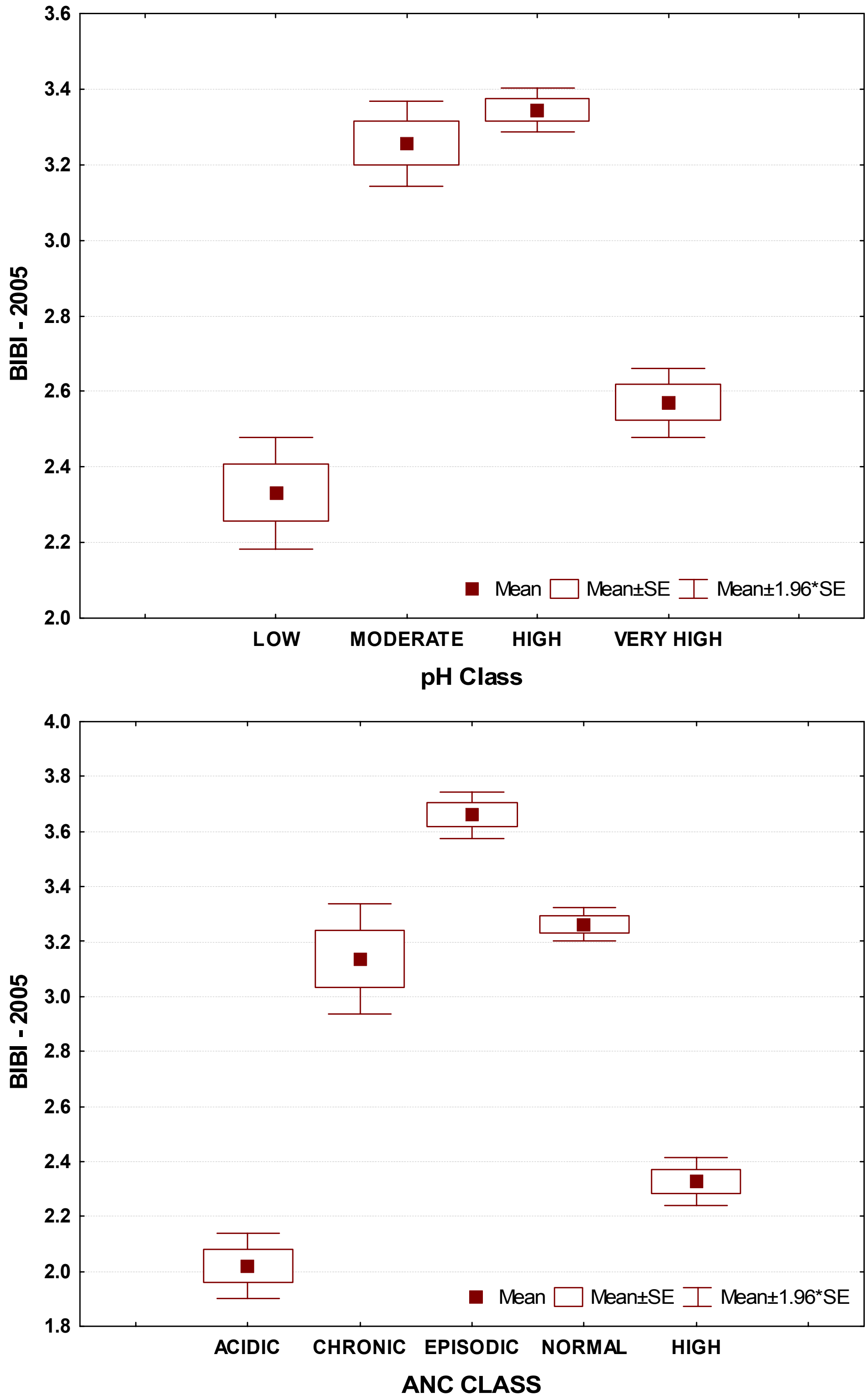


Figure 14-15. Summary of BIBI (2005) for four pH classes (upper) and five ANC classes (below) for MBSS sites. Classes are as described in Table 14-1 with the addition of very high pH > 7.5 and high ANC > 750 µeq/L classes.

Table 14-5. Fish species found at 2000-2004 MBSS sites by summer pH. *Species considered to be game species.				
Species	**pH < 5**	**pH 5.0-5.5**	**pH 5.5-6.0**	**pH 6.0-6.5**
ALEWIFE				X
AMERICAN EEL	X	X	X	X
BANDED KILLIFISH				X
BANDED SUNFISH	X	X	X	X
BLACK CRAPPIE		X	X	X
BLACKNOSE DACE		X	X	X
BLUE RIDGE SCULPIN			X	X
BLUEGILL		X	X	X
BLUESPOTTED SUNFISH	X	X	X	X
BLUNTNOSE MINNOW				X
BROOK TROUT*				X
BROWN BULLHEAD		X	X	X
BROWN TROUT*				X
CENTRAL STONEROLLER				X
CHAIN PICKEREL*		X	X	X
COMELY SHINER			X	
COMMON CARP				X
COMMON SHINER			X	X
CREEK CHUB		X	X	X
CREEK CHUBSUCKER		X	X	X
CUTLIPS MINNOW			X	X
EASTERN MUDMINNOW	X	X	X	X
FALLFISH			X	X
FANTAIL DARTER				X
FATHEAD MINNOW			X	X
FLIER			X	X
GLASSY DARTER				X
GOLDEN REDHORSE				X
GOLDEN SHINER		X	X	X
GOLDFISH				X
GREEN SUNFISH		X	X	X
GREENSIDE DARTER				X
IRONCOLOR SHINER				X
LARGEMOUTH BASS*		X	X	X
LEAST BROOK LAMPREY		X	X	X
LEPOMIS HYBRID				X
LONGNOSE DACE				X
MARGINED MADTOM		X	X	X
MOSQUITOFISH			X	X
MOTTLED SCULPIN				X
MUD SUNFISH	X	X	X	X
NORTHERN HOGSUCKER				X
PIRATE PERCH	X	X	X	X
POTOMAC SCULPIN				X
PUMPKINSEED	X	X	X	X

Table 14-5. (Continued)				
Species	**pH < 5**	**pH 5.0-5.5**	**pH 5.5-6.0**	**pH 6.0-6.5**
RAINBOW TROUT				**X**
REDBREAST SUNFISH		**X**	**X**	**X**
REDFIN PICKEREL	**X**	**X**	**X**	**X**
RIVER CHUB				**X**
ROCK BASS				**X**
ROSYSIDE DACE			**X**	**X**
SATINFIN SHINER			**X**	**X**
SEA LAMPREY			**X**	**X**
SMALLMOUTH BASS				**X**
SPOTFIN SHINER				**X**
SPOTTAIL SHINER			**X**	**X**
STRIPED BASS				**X**
SWALLOWTAIL SHINER			**X**	**X**
SWAMP DARTER			**X**	**X**
TADPOLE MADTOM			**X**	**X**
TESSELLATED DARTER			**X**	**X**
WARMOUTH			**X**	**X**
WHITE PERCH			**X**	
WHITE SUCKER			**X**	**X**
YELLOW BULLHEAD			**X**	**X**
YELLOW PERCH		**X**	**X**	**X**
TOTAL	**8**	**22**	**42**	**64**

found frequently in association with organically enriched acidic streams.

14.2.7 Fish Abundance Under Acidified or Acid Sensitive Conditions

The estimated density of fish (mean number of fish per stream mile) varied by species under acidified and acid-sensitive conditions (Table 14-6). Statewide estimates were calculated for the number of individual fish per stream mile within each of four ANC classes (< 0, 0-50, 50-200, and > 200 µeq/L). Estimates reported here were not adjusted for capture efficiency; it should be noted that the ANC measurement was done during the spring index and fish collection during the summer index period. Across all sites, the number of fish per stream mile declined with low ANC.

Thirty-seven fish species were found in all four of the ANC classes. Dramatic differences were seen in fish species composition and abundance above and below the threshold for acid sensitivity (ANC – 200 µeq/L). For many species, the highest number of stream miles fall into the highest ANC class, although 41 species were collected in the lowest ANC class (Table 14-6). Many species were collected in very low numbers so estimates of individual fish per stream mile are often less than zero.

Given that an estimated 27% of stream miles in the study area had an ANC less than 200 µeq/L, the effects of acidification on many fish populations appear to be significant. It is important to note that this analysis considered only acidification, not other natural (e.g., geographic) or anthropogenic effects on fish abundance. For example, brook trout tend to favor the high-gradient streams of western Maryland, where ANC conditions < 200 µeq/L are common. This geographic difference would explain the apparent increase in brook trout abundance in streams with ANC 50-200 µeq/L, compared to streams in other parts of the state that have ANC > 200 µeq/L but lack suitable habitat for brook trout.

Table 14-6. Mean number of fish per stream mile within each acid neutralizing capacity (ANC - µeq/L) class by species, 2000-2004 MBSS.

Species	ANC ≤ 0	SE	ANC 0 to 50	SE	ANC 50 to 200	SE	ANC > 200	SE
ALEWIFE	0	0	0	0	0	0	4	3
AMERICAN BROOK LAMPREY	0	0	0	0	0	0	2	1
AMERICAN EEL	7	2	8	2	47	8	143	19
BANDED KILLIFISH	0	0	0	0	1	0	10	5
BANDED SUNFISH	2	1	1	1	3	1	3	2
BLACK CRAPPIE	0	0	0	0	1	0	1	1
BLACKNOSE DACE	21	4	45	5	192	17	1,301	77
BLUE RIDGE SCULPIN	9	3	12	4	128	35	649	70
BLUEBACK HERRING	0	0	0	0	0	0	0	0
BLUEGILL	6	2	9	2	41	7	122	20
BLUESPOTTED SUNFISH	6	2	6	2	19	4	22	6
BLUNTNOSE MINNOW	6	4	13	8	24	5	667	285
BROOK TROUT	4	1	3	1	21	4	16	3
BROWN BULLHEAD	9	16	4	5	12	11	25	22
BROWN TROUT	0	0	0	0	2	1	24	10
CENTRAL STONEROLLER	2	1	5	3	32	10	100	25
CHAIN PICKEREL	1	0	1	0	3	1	3	0
CHANNEL CATFISH	0	0	0	0	0	0	0	0
CHECKERED SCULPIN	0	0	0	0	0	0	6	4
COMELY SHINER	0	0	0	0	0	0	1	1
COMMON CARP	0	0	0	0	0	0	0	0
COMMON SHINER	2	1	0	0	9	2	95	18
CREEK CHUB	12	3	15	5	64	10	292	22
CREEK CHUBSUCKER	3	1	6	1	26	5	40	8
CUTLIPS MINNOW	1	0	0	0	4	1	50	7
CUTTHROAT TROUT	0	0	0	0	0	0	0	0
CYPRINID (UNKNOWN)	0	0	0	0	0	0	0	0
CYPRINID HYBRID	0	0	0	0	0	0	0	0
EASTERN MUDMINNOW	71	14	77	13	460	72	750	176
EASTERN SILVERY MINNOW	0	0	0	0	0	0	7	6
FALLFISH	2	1	2	1	20	5	105	30
FANTAIL DARTER	4	2	9	4	35	11	184	39
FATHEAD MINNOW	0	0	0	0	1	1	8	3
FLIER	0	0	0	0	0	0	0	0
GIZZARD SHAD	0	0	0	0	0	0	2	1
GLASSY DARTER	0	0	0	0	0	0	3	3
GOLDEN REDHORSE	0	0	0	0	0	0	1	1
GOLDEN SHINER	6	3	6	2	30	8	39	8
GOLDFISH	0	0	0	0	0	0	0	0
GREEN SUNFISH	2	1	5	2	13	4	88	25
GREENSIDE DARTER	2	1	3	2	8	2	276	518
INLAND SILVERSIDE	0	0	0	0	0	0	0	0
IRONCOLOR SHINER	0	0	0	0	0	0	0	0
JOHNNY DARTER	0	0	1	0	2	1	1	0

Table 14-6. (Continued)								
Species	**ANC ≤ 0**	**SE**	**ANC 0 to 50**	**SE**	**ANC 50 to 200**	**SE**	**ANC > 200**	**SE**
LARGEMOUTH BASS	1	0	1	0	4	1	20	3
LEAST BROOK LAMPREY	3	1	9	3	39	14	81	38
LEPOMIS HYBRID	0	0	0	0	0	0	0	0
LOGPERCH	0	0	0	0	0	0	0	0
LONGEAR SUNFISH	0	0	0	0	1	0	1	0
LONGNOSE DACE	2	1	3	1	38	13	231	28
MARGINED MADTOM	1	1	1	1	17	10	61	35
MOSQUITOFISH	1	1	2	1	7	5	30	12
MOTTLED SCULPIN	9	2	9	3	44	12	13	4
MUD SUNFISH	0	0	0	0	0	0	0	0
MUMMICHOG	0	0	0	0	1	1	21	10
NORTHERN HOGSUCKER	0	0	1	0	3	1	25	6
NOTROPIS SP.	0	0	0	0	0	0	0	0
PEARL DACE	0	0	1	1	0	0	14	11
PIRATE PERCH	10	4	10	4	60	22	88	84
POTOMAC SCULPIN	1	0	2	1	21	6	140	38
PUMPKINSEED	5	2	4	1	23	4	65	14
RAINBOW DARTER	0	0	0	0	17	7	10	3
RAINBOW TROUT	0	0	0	0	0	0	1	0
REDBREAST SUNFISH	1	0	2	1	14	2	67	10
REDFIN PICKEREL	6	2	7	3	29	9	21	4
RIVER CHUB	0	0	0	0	26	17	105	68
ROCK BASS	0	0	1	0	10	3	10	2
ROSYFACE SHINER	0	0	0	0	2	1	18	9
ROSYSIDE DACE	5	1	3	1	25	4	267	25
SATINFIN SHINER	1	1	0	0	4	1	35	9
SEA LAMPREY	0	0	0	0	4	2	30	14
SHIELD DARTER	0	0	0	0	0	0	6	2
SHORTHEAD REDHORSE	0	0	0	0	0	0	0	0
SILVERJAW MINNOW	1	1	2	2	0	0	53	33
SMALLMOUTH BASS	0	0	0	0	9	2	21	4
SPOTFIN SHINER	0	0	0	0	7	3	41	21
SPOTTAIL SHINER	2	1	3	2	2	1	260	192
STRIPED BASS	0	0	0	0	0	0	0	0
STRIPED SHINER	0	0	0	0	0	0	0	0
SUNFISH (UNKNOWN)	0	0	0	0	0	0	0	0
SUNFISH HYBRID	0	0	0	0	0	0	0	0
SWALLOWTAIL SHINER	1	1	1	0	6	2	90	20
SWAMP DARTER	0	0	0	0	1	0	0	0
TADPOLE MADTOM	2	1	2	2	11	8	7	7
TESSELLATED DARTER	18	9	9	3	77	12	429	56
WARMOUTH	0	0	1	0	2	1	1	1
WHITE CATFISH	0	0	0	0	0	0	0	0
WHITE PERCH	0	0	0	0	0	0	0	0

Table 14-6. (Continued)								
Species	**ANC ≤ 0**	**SE**	**ANC 0 to 50**	**SE**	**ANC 50 to 200**	**SE**	**ANC > 200**	**SE**
WHITE SUCKER	7	1	5	1	31	6	280	27
YELLOW BULLHEAD	1	1	1	1	5	2	21	9
YELLOW PERCH	0	0	0	0	0	0	1	1
Species Number	**41**		**42**		**57**		**69**	

14.3 NUTRIENTS

This section presents water quality results relating to nutrient and dissolved oxygen concentrations from the 1995-1997 and 2000-2004 MBSS. Levels of nitrate (as NO_3-N), nitrite (as NO_2-N), ammonia (as NH_3-N), total nitrogen, total phosphorus, ortho-phosphate, and dissolved oxygen (DO) are examined for first- (N = 845), second- (N = 265) and third-order (N = 140) streams in each basin sampled in the 2000-2004 MBSS (in the second MBSS round, only 39 fourth-order sites were sampled). In addition, the 1995-1997 MBSS sampled 325 first-order, 332 second-order, and 297 third-order streams for water quality for a total of 2243 streams sampled over eight years.

14.3.1 Background Information on Nutrients

In aquatic systems, nutrients such as nitrogen and phosphorus (and major elements such as carbon, oxygen and sulfur and trace elements such as calcium, magnesium, potassium, sodium, etc.) are essential for life (Allan 1995). Without human influence, streams contain background levels of nitrogen and phosphorus essential to the survival of autotrophic, and also heterotrophic, organisms within that basin. However, over the last four hundred years, nutrient loading in many stream systems increased, resulting from anthropogenic influences such as agricultural runoff, wastewater discharge, atmospheric deposition, and urban/suburban point and non-point sources (Galloway et al. 2003). Increased sediment loading and elevated stream temperatures affect nutrient processing in aquatic systems (Walters 1995).

Elevated nitrogen concentrations contribute to nutrient enrichment in aquatic systems, with reactive nitrogen accumulating at all spatial scales (Galloway et al. 2003). Excessive nitrogen loading may lead to the eutrophication of water bodies, particularly in lakes or downstream estuaries. Eutrophication often decreases the level of dissolved oxygen available to aquatic organisms, with a prolonged exposure to low dissolved oxygen values suffocating adult fish or leading to reduced recruitment. Increased nutrient loads are also thought to be harmful to humans by causing toxic algal blooms and contributing to outbreaks of toxic organisms such as *Pfiesteria piscicida* in the Chesapeake Bay. Besides contributing to a loss of biodiversity and habitat degradation in coastal ecosystems, reactive nitrogen also contributes to impacts on human and ecosystem health (Galloway et al. 2003). There is a constant biotic struggle to capture phosphorus, an element that is limiting in freshwater systems (Allan 1995). Although the phosphorus cycle is not nearly as complex as stream nitrogen dynamics, it is important to consider in freshwater systems, where excessive total phosphorus may be present, since it may accelerate eutrophication, and alter stream processing of materials.

In Maryland, concern for nutrient loadings to the Chesapeake Bay has drawn attention to the amounts of materials transported throughout the basin by stream tributaries. In the Chesapeake Bay basin, the largest source of nitrogen is from agriculture (estimated as 39% of total nitrogen). Other contributors include point sources (23%), runoff from developed areas (9%) and forests (18%), and direct atmospheric deposition to the Chesapeake Bay (11%). The total contribution of atmospheric deposition is higher (27%), including amounts deposited to the basin and subsequently entering the Chesapeake Bay as runoff (Chesapeake Bay Program 1995). Atmospheric deposition is therefore recognized as a significant contributor of nitrogen to the Chesapeake Bay, including deposition reaching the basin from power plants and other distant sources within the airshed (Dennis 1996).

The MBSS provides a large dataset for assessing nutrient concentrations under spring baseflow conditions. Although a full understanding of nutrient loadings also requires data collected over multiple years and seasons, the MBSS water chemistry results provide extensive spatial coverage (with nearly 2250 sites sampled), enabling nutrient concentrations to be compared among basins statewide. In the 2000-2004 MBSS, the more complete set of nutrient analyses allows for a more intensive and extensive assessment.

14.3.2 Results of Nutrient Assessment

In the 2000-2004 MBSS, concentrations of several nutrients were determined during the spring baseflow period, as well as summer stream dissolved oxygen (Table 14-7). Thresholds for these nutrients and dissolved oxygen were estimated by examining statistical

Table 14-7. MBSS water quality thresholds for nutrients and dissolved oxygen measured in 2000-2004 MBSS. Both nitrate and dissolved oxygen were also measured in 1995-1997. All units are in mg/l.			
Parameter	**Low**	**Moderate**	**High**
Nitrate–N	< 1.0	1.0 – 5.0	> 5.0
Nitrite–N	< 0.0025	0.0025 – 0.01	> 0.01
Ammonia-N	< 0.03	0.03 – 0.07	> 0.07
TN	< 1.5	1.5 – 7.0	>7.0
TP	< 0.025	0.025 – 0.070	> 0.070
Othro-PO_4	< 0.008	0.008 – 0.03	> 0.03
Dissolved Oxygen	< 3	3-5	> 5

distributions of each nutrient, employing basins with greater than 90% forest as a reference – the thresholds for dissolved oxygen are identical to those used in 1995-1997. For all box and whisker plots, the absence of a mean box indicates that no samples were collected for that basin-stream order combination; a mean box without the mean ± SE box or whiskers indicates that a single sample was collected for that basin-stream order combination. For the ten Maryland tributary strategy basins (excluding the Youghiogheny and Ocean Coastal basins), nutrient analyses were broken down by seven basin areas: < 500 acres (202 ha); 500-1,000 acres (202-405 ha); 1,000-2,500 acres (405–1,012 ha); 2,500-5,000 acres (1,012-2,024 ha); 5,000-10,000 acres (2,024-4,047 ha); 10,000-20,000 acres (4,047-8,094 ha); and greater than 20,000 acres (>8,094 ha).

14.3.2.1 Nitrate

The majority of the MBSS basins, by stream order, have nitrate concentrations (all nitrate values measured as nitrate-nitrogen) greater than the threshold value between low (< 1.0 mg/l) and moderate (1-5 mg/l) nitrate concentrations (Table 14-7 and Figure 14-16). However, no basin had a mean nitrate concentration greater than 5 mg/l (Figure 14-16). For first-order MBSS streams, 14 basins had mean nitrate levels greater than 1 mg/l, with only the North Branch of the Potomac River, West Chesapeake, and Youghiogheny basins being less than the nitrate threshold value. In second-order MBSS streams, 14 basins had moderate nitrate levels and four basins had nitrate levels less than 1.0 mg/l (Figure 14-16), paralleling the same pattern seen in the third-order streams for the same basins (only one third-order stream sampled in the Nanticoke-Wicomico basin). Limited nitrate data exist for fourth-order streams in the MBSS, but the Gunpowder, Middle Potomac, Patuxent and Lower Susquehanna exceeded the 1.0 mg/l threshold (single point estimates not included). Three basins had mean nitrate levels greater than 4.0 mg/l – these include the second-order streams in the Nanticoke-Wicomico basin and third-order streams in the Chester and Lower Susquehanna basins (Figure 14-16). For all first- through third-order streams, there are four basins (Lower Potomac, North Branch, West Chesapeake, and Youghiogheny) with mean nitrate concentrations less than 1.0 mg/l (the North Branch and the Upper Potomac nitrate values are less than 1.0 mg/l for fourth-order streams).

Nitrate-nitrogen was measured in both MBSS rounds so comparisons may be made of nitrate concentration over time by 8 digit basin (Table 14-8 and Figure 14-18). There was a significant reduction in stream nitrate concentration from the first (2.4 mg/l) to the second round (1.8 mg/l), as reflected in the difference in median values (Table 14-8). This nitrate difference is better illustrated in Figure 14-17, where there is a significant difference in nitrate levels for the first- and third-order streams for each of the two MBSS rounds. Analysis by basin (Ocean Coastal was not sampled in 1995-1997 MBSS) for each MBSS round also reveals the same nitrate pattern (Figure 14-18). There were general declines in mean nitrate levels in all basins that had elevated mean nitrate concentrations (note that the North Branch, West Chesapeake, and Youghiogheny basins were less than 1.0 mg/l). Reasons for these decreases in nitrate may be related to decreased atmospheric nitrogen deposition, a reduction in fertilizer use (perhaps due to improved cover crop management, or implemented agricultural nutrient management plans with BMPs), remediation of septic systems, or restoration of riparian corridors (CREP program and others). They may also be an artifact of the greater number of small streams sampled in 2000-2004.

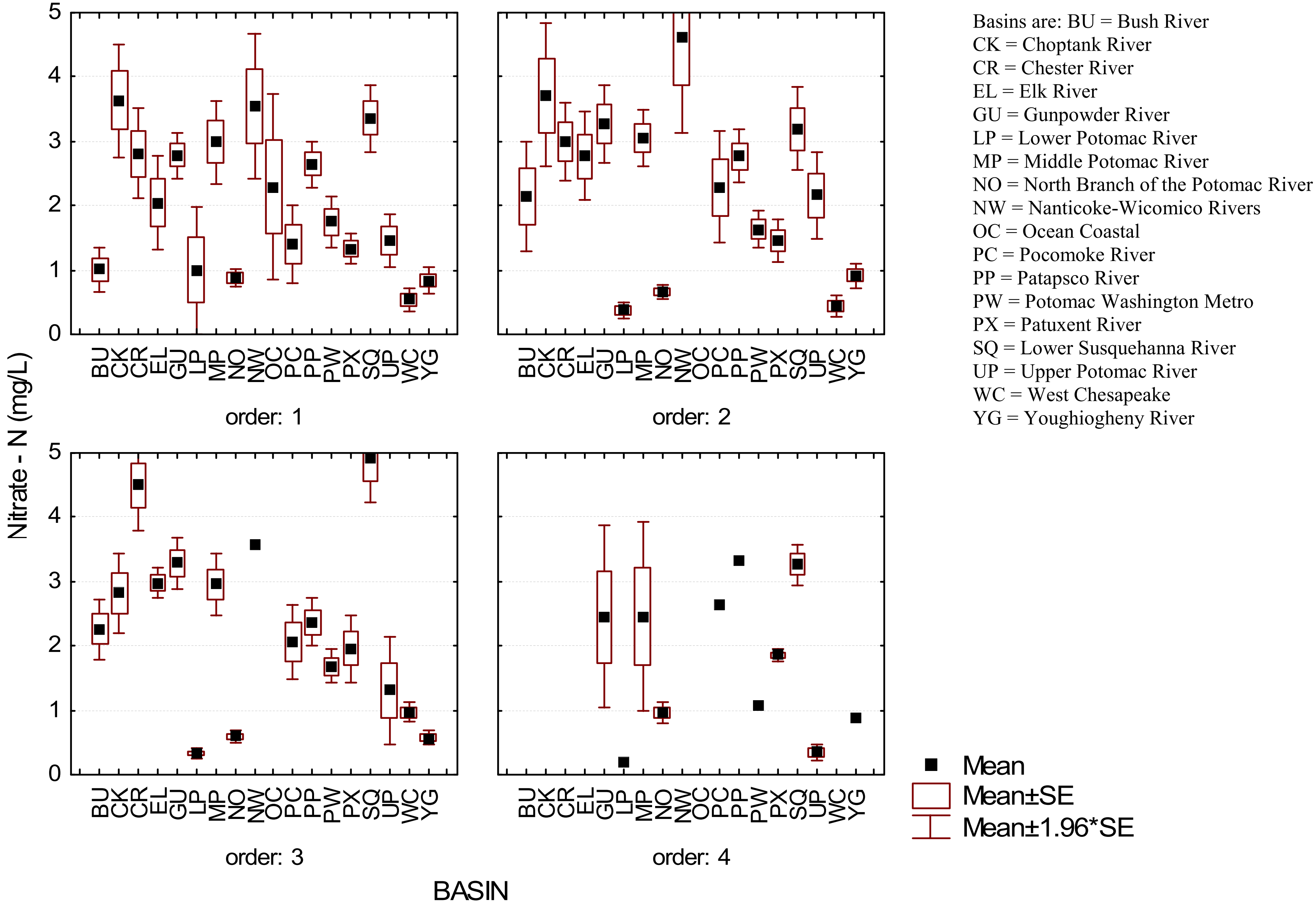


Figure 14-16. Nitrate-nitrogen (mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS

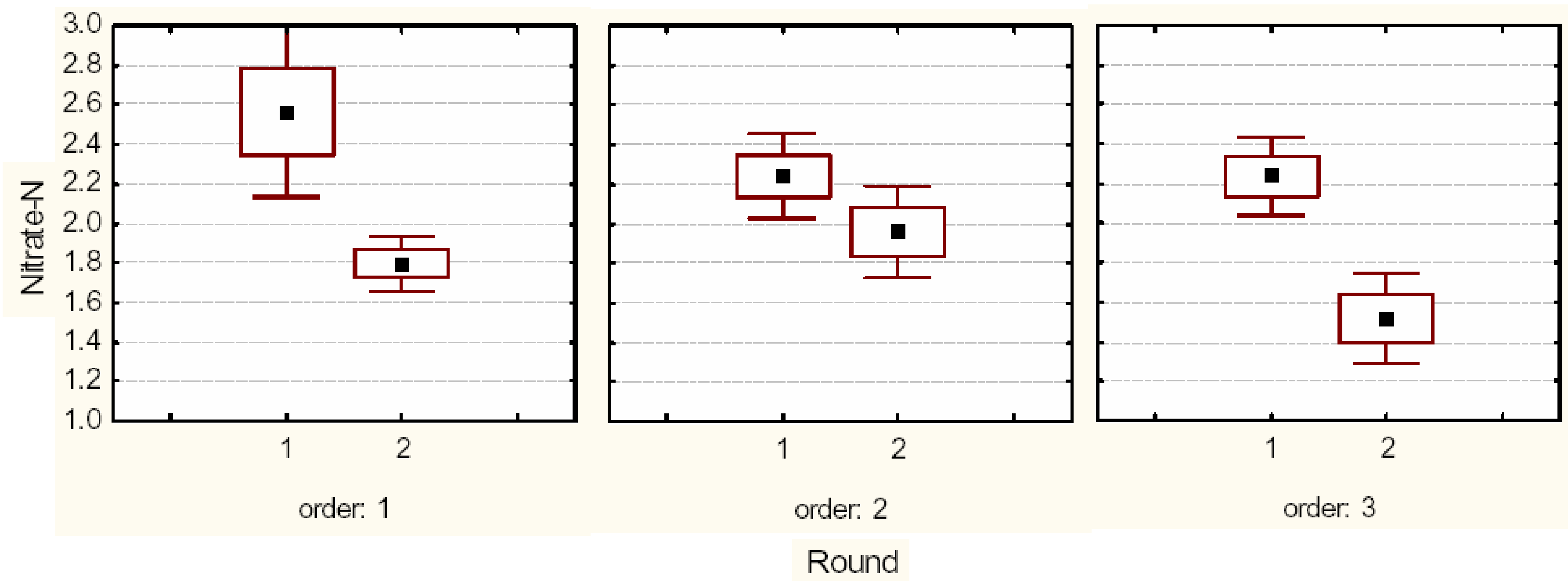
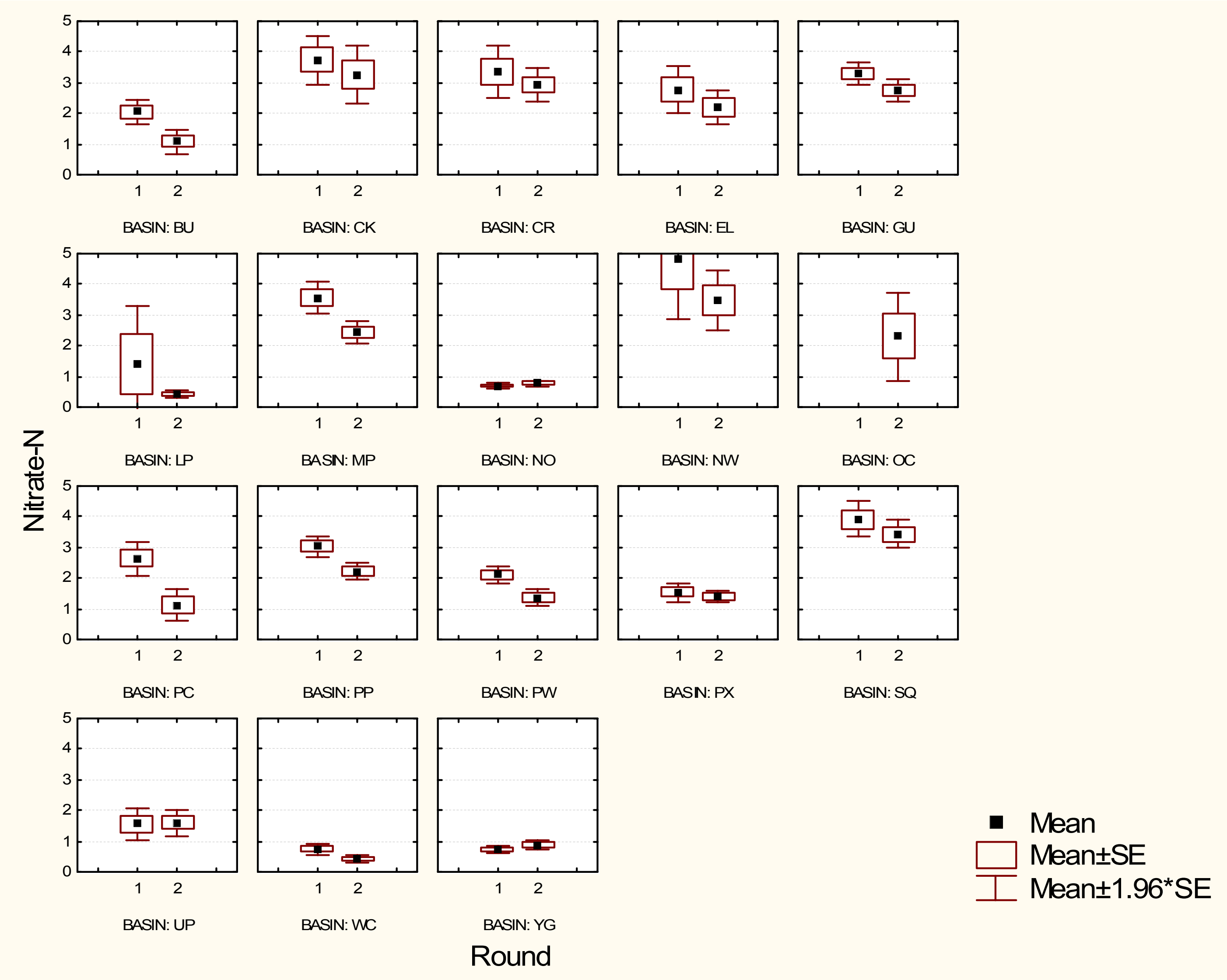


Figure 14-17. Nitrate-nitrogen (mg/l) concentration by MBSS round (Round 1 and Round 2) for stream order (1-3).



Basins are: BU = Bush River; CK = Choptank River; CR = Chester River; EL = Elk River; GU = Gunpowder River; LP = Lower Potomac River; MP = Middle Potomac River; NO = North Branch of the Potomac River; NW = Nanticoke-Wicomico Rivers; OC = Ocean Coastal; PC = Pocomoke River; PP = Patapsco River; PW = Potomac Washington Metro; PX = Patuxent River; SQ = Lower Susquehanna River; UP = Upper Potomac River; WC = West Chesapeake; and YG = Youghiogheny River.

Figure 14-18. Nitrate-nitrogen (mg/l) concentration by MBSS round (Round 1 and Round 2) for all basins.

Table 14-8. Summary of nutrient parameters for MBSS water quality collected during spring baseflow in first- through third-order streams (Round One = 1995-1997; Round Two = 2000-2004); all units = mg/l

Parameter	Mean	Median	SE	25% Percentile	75% Percentile	N
Nitrate-N	2.06	1.35	0.050	0.52	2.98	2190
Round One	2.36	1.65	0.089	0.63	3.38	949
Round Two	1.82	1.14	0.055	0.41	2.68	1241
Nitrite-N	0.0078	0.0046	0.00033	0.0016	0.0097	1140
Ammonia-N	0.049	0.015	0.0048	0.0065	0.039	1212
Total N	2.09	1.40	0.057	0.65	3.01	1266
Total P	0.041	0.020	0.0022	0.011	0.043	1266
Ortho-PO_4	0.012	0.0037	0.0015	0.00070	0.0088	1232

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, four basins exceeded 10% of stream km for the high nitrate-nitrogen threshold of 5 mg/l (Table 14-9). The Lower Eastern Shore and the Choptank had greater than 15% of stream km above the high nitrate threshold, while all basins had greater than 50% of stream km above the moderate threshold. The Lower Eastern Shore, Lower Western Shore, and the Lower Potomac had the highest percentages (14.3 – 34.7) of stream km in the low category (Table 14-9).

In the 1995-1997 MBSS, the majority of stream km (59%) had NO_3-N concentrations greater than 1.0 mg/l. An estimated 41% of stream km had NO_3-N concentrations between 0.1 mg/l and 1.0 mg/l, and only 0.4% had concentrations that were less than 0.1 mg/l. Only three basins had any stream km (< 5%) with less than 0.1 mg/l of NO_3-N: the Upper Potomac, the Lower Potomac, and the West Chesapeake. An estimated 29% of stream km had a NO_3-N concentration greater than 3.0 mg/l and an estimated 5% of stream km had a NO_3-N concentration greater than 7.0 mg/l. Areas where the concentration is greater than 7.0 mg/l are places where NO_3-N may be especially detrimental to stream quality. These areas occurred in seven of the basins sampled: Upper Potomac, Middle Potomac, Lower Potomac, Patuxent and Patapsco (1995 and 1997 sampling).

Of 2190 nitrate-nitrogen measurements made in both MBSS rounds, only 0.87% (N = 19) exceeded the U. S. Environmental Protection Agency (EPA) Maximum Contaminant Level of 10 mg/l of nitrate-nitrogen (criteria for ground water and drinking water). The highest stream nitrate measured was 52.7 mg/l.

14.3.2.2 Nitrite

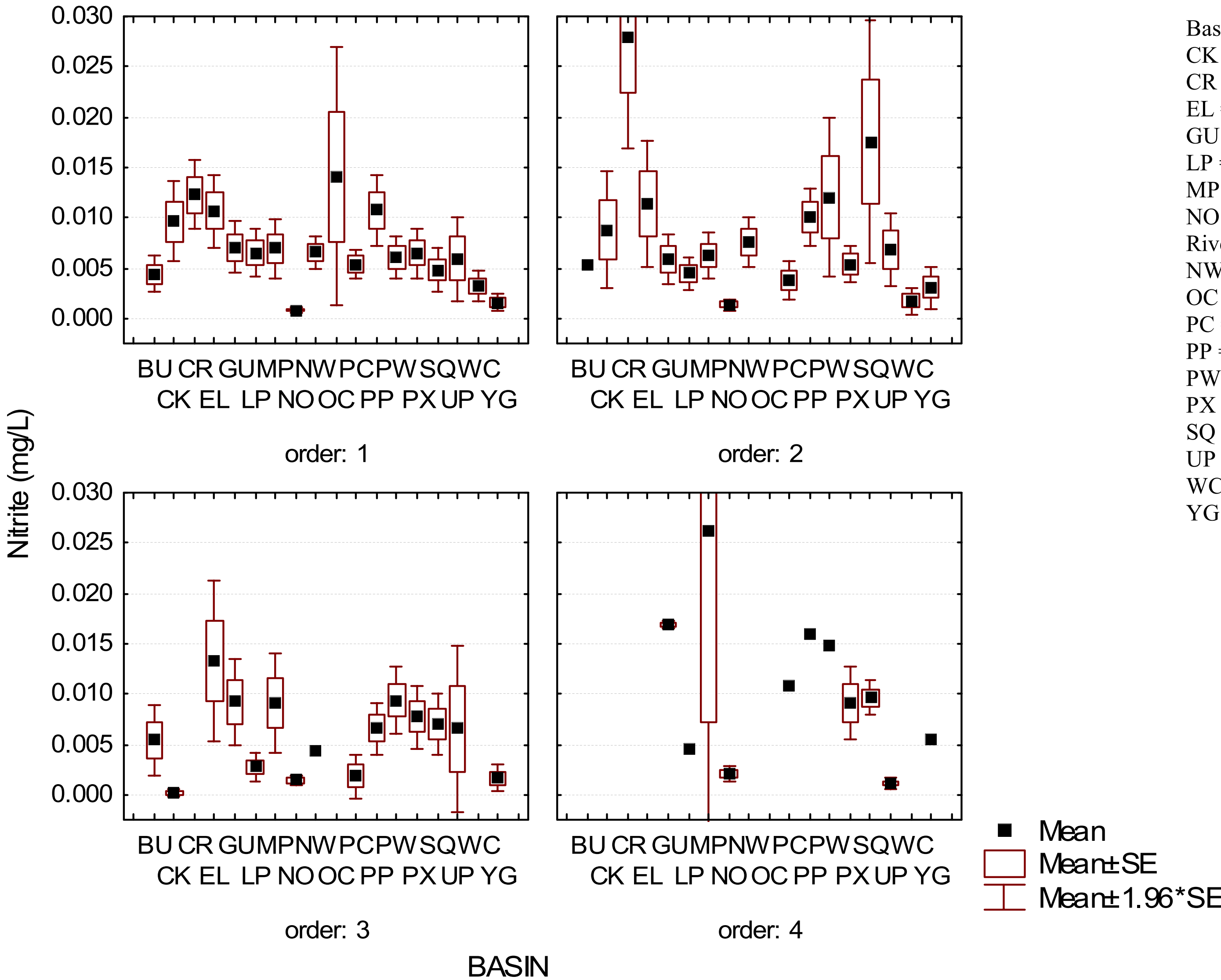
Nitrite (as nitrite-nitrogen) was first measured in the 2000-2004 MBSS, with values for many basins, by stream order, exceeding the low/moderate cutoff threshold of 0.0025 mg/l (Table 14-7). Five basins for first-order streams, six basins for second-order streams, three basins for third-order streams, and three fourth-order streams (only those with adequate sample size considered) exceeded the upper threshold of 0.010 mg/l (Figure 14-19). Of the 18 MBSS basins, only the North Branch and Youghiogheny had mean nitrite values less than 0.0025 mg/l in the first-order streams. For second-order streams, only the North Branch and West Chesapeake basins were below the 0.0025 mg/l threshold: for third-order streams, the North Branch, Pocomoke West Chesapeake and Youghiogheny basins had low nitrite levels. A limited number of fourth-order streams were sampled in the second MBSS round, but for those basins with adequate sample sizes (GU, MP, NO, PX, SQ, and UP – Figure 14-19), only the North Branch (NO) and the Upper Potomac (UP) were below the 0.0025 mg/l nitrite threshold.

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, seven basins exceeded the high nitrite-nitrogen threshold of 0.01 mg/l with greater than 20% of stream km (Table 14-9). Two basins had > 50% of stream km in the moderate threshold. The Youghiogheny (81%) and Lower Western Shore (63%) had the highest percentages of stream km in the low threshold category (Table 14-9).

Of 1,140 nitrite-nitrogen measurements made in the 2000-2004 MBSS, no samples exceeded the U.S. Environmental Protection Agency Maximum Contaminant Level of 1 mg/l of nitrite-nitrogen (criteria for ground water and drinking water). The highest stream nitrite measured was 0.15 mg/l, with an overall mean nitrite of 0.0078 mg/l (Table 14-8).

Table 14-9. Percentage of stream km within each threshold category as defined in Table 14-7 for nutrients and dissolved oxygen for all tributary strategy basins. The Youghiogheny and Ocean Coastal basins are included but their drainages are not part of the Chesapeake Bay basin.

		Basin											
Analyte (mg/l)	**Threshold**	**Lower Eastern Shore**	**Choptank**	**Upper Eastern Shore**	**Upper Western Shore**	**Patapsco/ Back River**	**Lower Western Shore**	**Patuxent**	**Lower Potomac**	**Middle Potomac**	**Upper Potomac**	**Youghiogheny**	**Ocean Coastal**
Nitrate-N	Low	14.3	4.0	4.4	2.2	1.2	17.1	7.0	34.7	3.8	5.4	1.7	
	Moderate	65.8	67.6	87.1	87.8	92.6	82.9	93.0	65.3	90.3	90.2	98.3	88.9
	High	19.9	28.4	8.6	10.0	6.2				5.8	4.5		11.1
Nitrite-N	Low	26.2	33.7	9.5	21.7	13.5	62.7	31.0	37.9	19.2	54.9	80.7	
	Moderate	53.9	23.6	39.8	53.3	41.5	34.3	52.1	49.2	48.6	26.9	19.3	77.8
	High	19.9	42.7	50.7	25.0	45.0	3.0	16.9	12.9	32.2	18.2		22.2
Ammonia-N	Low	58.9	56.8	45.4	83.4	65.9	47.4	39.1	63.6	73.6	88.0	94.1	77.8
	Moderate	24.8	7.1	28.5	7.1	17.9	33.6	42.4	23.3	11.1	3.7	4.3	11.1
	High	16.3	36.2	26.1	9.6	16.2	19.0	18.5	13.1	15.2	8.3	1.7	11.1
TN	Low	46.9	15.6	20.8	14.1	27.5	96.4	65.3	89.0	46.4	49.9	82.1	33.3
	Moderate	39.0	65.7	72.5	81.8	71.5	3.6	34.7	11.0	51.6	47.0	17.9	55.6
	High	14.1	18.7	6.8	4.1	1.0				1.9	3.1		11.1
TP	Low	47.0	9.1	12.3	65.9	72.5	24.5	34.7	62.2	60.4	68.6	87.9	11.1
	Moderate	28.3	64.4	45.4	25.4	19.2	47.0	35.1	28.2	34.5	20.9	12.1	44.4
	High	24.7	26.5	42.3	8.8	8.3	28.4	30.2	9.7	5.1	10.6		44.4
Ortho-PO_4	Low	55.7	33.3	44.8	67.6	76.3	77.0	74.7	76.5	83.4	73.0	100	11.1
	Moderate	29.6	46.1	38.7	26.9	19.8	23.0	21.9	23.5	11.1	15.8		22.2
	High	14.7	20.5	16.6	5.6	3.9		3.4		5.5	11.2		66.7
DO	Low	21.4	16.4	7.0	1.2	2.1	7.5	2.0	10.8	0.5			
	Moderate	27.2	8.9	8.3	8.3	2.4	12.3	1.6	8.4	2.5	3.3	4.3	14.3
	High	51.3	74.8	84.7	90.5	95.5	80.2	96.4	80.8	96.9	96.7	95.7	85.7



Basins are: BU = Bush River
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GU = Gunpowder River
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MP = Middle Potomac River
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NW = Nanticoke-Wicomico Rivers
OC = Ocean Coastal
PC = Pocomoke River
PP = Patapsco River
PW = Potomac Washington Metro
PX = Patuxent River
SQ = Lower Susquehanna River
UP = Upper Potomac River
WC = West Chesapeake
YG = Youghiogheny River

Figure 14-19. Nitrite-nitrogen (mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.

14.3.2.3 Ammonia

Ammonia (as ammonia-nitrogen) was also measured in the 2000-2004 MBSS; values for ammonia for all basins and all four stream orders often exceeded the low/moderate cutoff concentration of 0.03 mg/l (Table 14-7 and Figure 14-20). For first-order streams, 13 basins exceeded the low threshold; five basins were below the low ammonia threshold (Figure 14-20). Eleven basins had ammonia levels above the threshold for second-order streams, with only three third-order streams exceeding the low threshold. For fourth-order streams (with adequate samples size), only the Middle Potomac exceeded the low ammonia threshold. There were several basins that exceeded the high cutoff (> 0.07) for ammonia; five basins were greater than the threshold for first-order streams, one for second order, none for third-order, and one – the Middle Potomac – for fourth-order streams. Of 1,212 ammonia-nitrogen measurements made in the 2000-2004 MBSS, the highest stream ammonia value measured was 2.8 mg/l, with overall mean ammonia of 0.049 mg/l (Table 14-8).

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, seven basins had > 15% of stream km in the high ammonia threshold of 0.07 mg/l, with nine basins having greater than 50% of stream km in the low threshold category (Table 14-9). The Upper Potomac and Youghiogheny basins had > 80% of their stream km in the low category.

14.3.2.4 Total Nitrogen

Total nitrogen (TN), first measured in the 2000-2004 MBSS, varied over all basins for all stream orders (Figure 14-21). Eleven basins, for first-order streams, exceeded the TN threshold of 1.5 mg/l, with twelve basins (second-order streams) also greater than the low TN threshold. Eleven third-order stream basins exceeded the TN threshold, along with four fourth-order basins (Figure 14-21). No basin, for any stream order, was greater than the high threshold of 7.0 mg/l TN, although several basins were obviously elevated for TN (Figure 14-21). Of 1266 TN measurements made in the second MBSS round, the highest stream value measured was 15.5 mg/l, with an overall mean TN of 2.09 mg/l (Table 14-8).

In December 2000, the U.S. EPA published ambient water quality criteria recommendations (TP, TN, chlorophyll *a*, and turbidity) for rivers and streams in Nutrient Ecoregions (aggregated ecoregions throughout the United States). There are three Nutrient Ecoregions associated with Maryland (U.S. EPA 2000a, 2000b, 2000c); Nutrient Ecoregion IX is the Southeastern Temperate Forested Plains and Hills (equivalent to sections of the western Coastal Plain and the entire Piedmont), Nutrient Ecoregion XI is the Central and Eastern Forested Uplands (equivalent to the Blue Ridge, Ridge and Valley, and Allegheny Plateau), and Nutrient Ecoregion XIV is the Eastern Coastal Plain (equivalent to the Coastal Plain on the Eastern Shore, and a section of the western Coastal Plain). The TN criteria for Nutrient Ecoregion IX is 0.69 mg/l, XI 0.31 mg/l, and XIV 0.71 mg/l; all below the MBSS threshold of 1.5 mg/l.

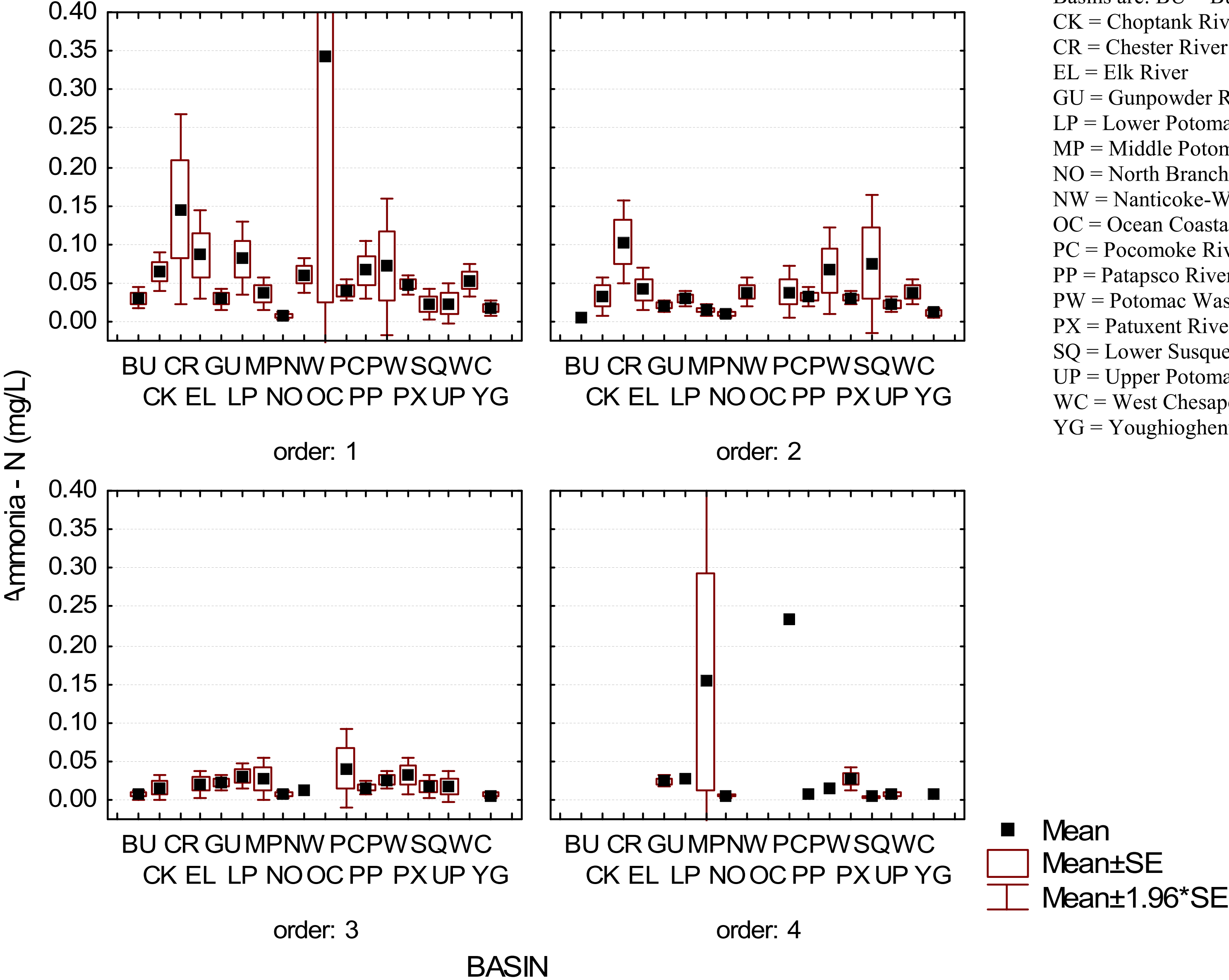
For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, three basins exceeded 10% of stream km for the high total nitrogen threshold of 7.0 mg/l (Table 14-9). Several basins had high percentages (> 50%) of stream km in the moderate (1.5 – 7.0 mg/l) category, including the Choptank, Upper Eastern Shore, Upper Western Shore, Middle Potomac and Ocean Coastal. Only two basins, Lower Western Shore and Youghiogheny, had greater than 80% of stream km in the low threshold category (Table 14-9).

14.3.2.5 Total Phosphorus

Total phosphorus (TP), measured in the 2000-2004 MBSS, also varied over all basins for all stream orders (Figure 14-22). Several basins had mean TP levels over 0.025 mg/l. Fifteen basins (first-order streams) exceeded the low TP threshold, with only the Lower Susquehanna, Youghiogheny, and North Branch below the threshold value. Fourteen basins (second-order streams) were greater than the threshold value, with nine basins (third-order) also higher than the low TP threshold level. The Patuxent, for fourth-order streams, was greater than 0.025 mg/l TP (Figure 14-22). For all basin-stream order combinations, six basins had TP greater than 0.07 mg/l (the high threshold). In particular, the Ocean Coastal (first-order), Chester (second-order), and Middle Potomac (fourth-order) all were higher than 0.10 mg/l TP.

Of 1266 TP measurements made in the MBSS, the highest value measured was 1.52 mg/l, with an overall mean TP of 0.041 mg/l (Table 14-8). The TP criteria for Nutrient Ecoregion IX is 0.037 mg/l, XI 0.010 mg/l, and XIV 0.031 mg/l; these criteria are close to the low TP threshold in Table 14-7 except for the criteria for Nutrient Ecoregion XI (0.010 mg/l versus 0.070 mg/l).

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, six basins exceeded 20% of stream km for the high total phosphorus threshold (Table 14-9). Three basins, in the moderate category, had stream km greater than 40%. One basin, the Youghiogheny, had greater than 80% of stream km in the low threshold category.



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YG = Youghiogheny River.

Figure 14-20. Ammonia (mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.

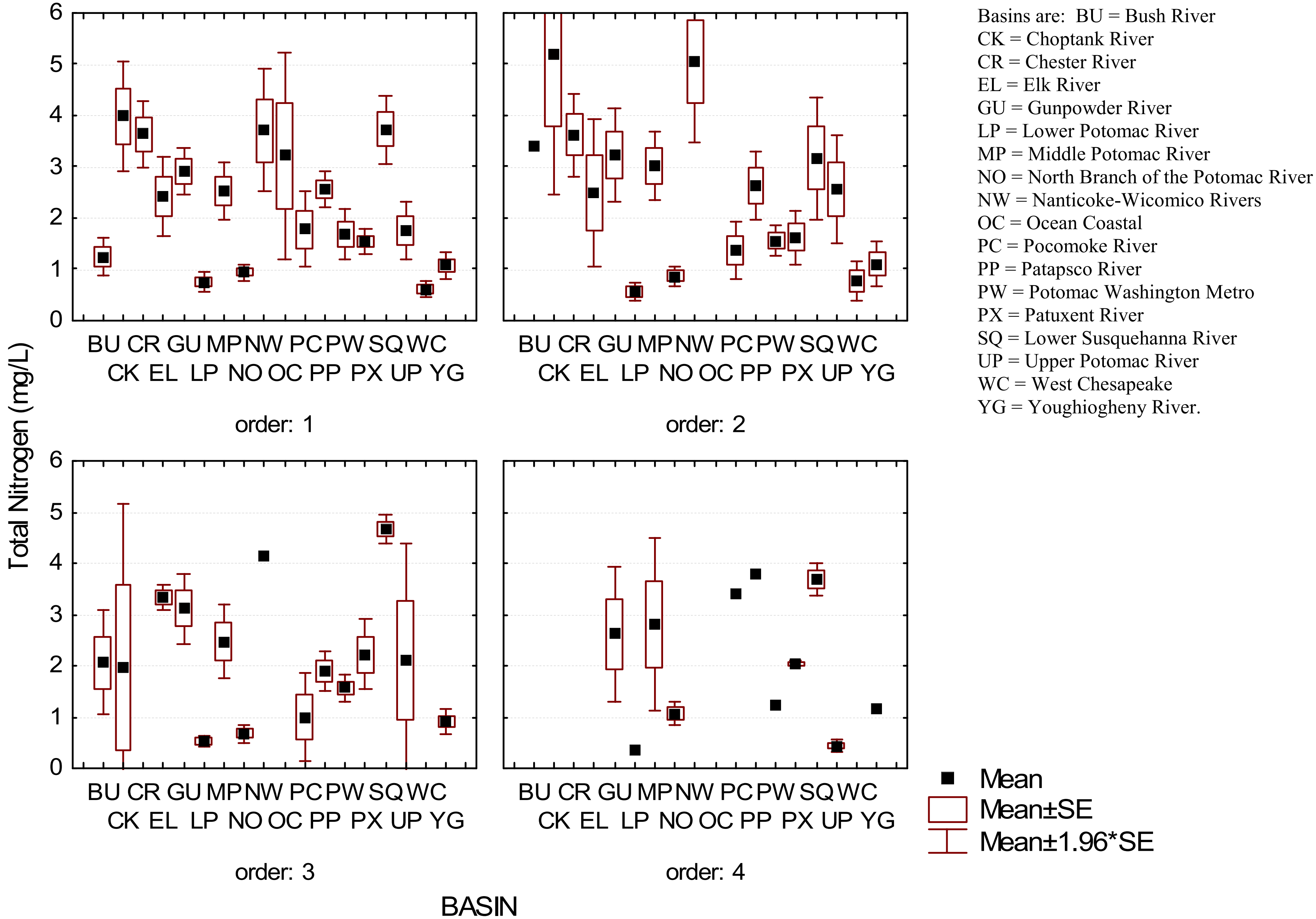
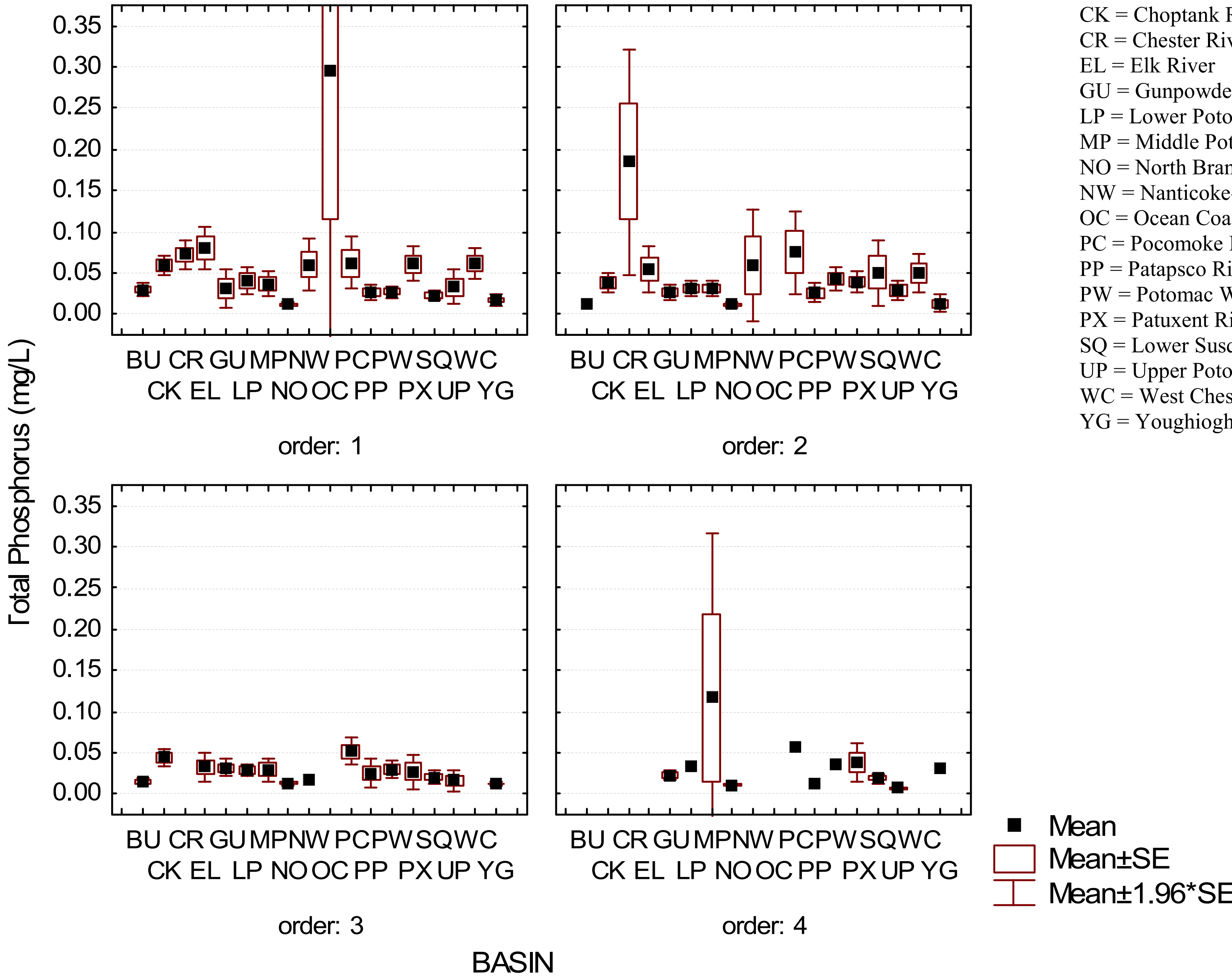


Figure 14-21. Total nitrogen (TN - mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.



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YG = Youghiogheny River.

Figure 14-22. Total phosphorus (TP - mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.

14.3.2.6 Ortho-phosphate

Ortho-phosphate (OP_4), measured in the 2000-2004 MBSS, again varied over all basins for all stream orders (Figure 14-23). The first-order Ocean Coastal streams are not plotted in Figure 14-26 since the mean OP_4 was 0.22 mg/l; the OP_4 for the Middle Potomac was also not plotted since the mean was 0.071 mg/l. Twenty-five basin-stream order combinations had OP levels > 0.008 mg/l, with many basins greater than the upper threshold of 0.03 mg/l for high OP_4. Of 1232 OP_4 measurements made in the second MBSS round, the highest value measured was 1.20 mg/l, with an overall mean OP_4 of 0.012 mg/l (Table 14-8).

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, five basins exceeded 10% of stream km for the high OP_4 threshold of 0.03 mg/l (Table 14-9), with four basins having 25% of stream km in the moderate OP_4 category. Four basins had greater than 75% of stream km in the low threshold category.

14.3.2.7 Dissolved Oxygen (DO)

Summer field DO normally exceeded the low threshold (Table 14-7) for all basin-stream order combinations (Figure 14-24). Only 4.3% of the 2176 summer DO measurements were below 3 mg/l, 6% between 3-5 mg/l, and 89.7% greater than 5.0 mg/l. Mean values for several basins, by stream order, were above 7 mg/l. The plot of field temperature versus DO (Figure 14-25) indicated that there is a subset of MBSS stations, with a field DO of less than 3 mg/l and a field temperature between 15-25 ºC, where the normal DO-temperature relationship is not present. Many of these stations are associated with basins having high nutrient loadings.

For all Chesapeake Bay tributary strategy basins, and the Youghiogheny and Ocean Coastal basins, there were three basins, Lower Eastern Shore, Choptank, and Lower Potomac, with greater than 10% of stream km in the low oxygen category of < 3 mg/l DO. All basins had > 50% of stream km in the high threshold category of > 5 mg/l DO, while five basins exceeded 90% (Table 14-9).

14.3.3 Tributary Strategy Basins

While nutrient increases can have deleterious effects on streams, eutrophication is often a bigger problem for receiving estuaries. For Maryland, there are ten large basins defined as Tributary Strategy Basins, essentially waters that eventually flow into tidal systems of the Chesapeake Bay. One large western Maryland basin, the Youghiogheny, flows into the Ohio basin, and eventually to the Mississippi River. A second drainage basin, the Ocean Coastal, drains into the Atlantic Ocean through tidal reaches of inland bays on the Atlantic Coast of Maryland.

Both TN and TP mean values were calculated for all ten Tributary Strategy Basins (Figures 14-26 and 14-27). For TN, there are three distinct groupings (Figure 14-26). The Lower Potomac and the Lower Western Shore had mean TN levels less than 1.0 mg/l (approximate to two eco-region values of 0.69 and 0.71 mg/l TN), indicating low TN in these two basins. A second grouping of the Upper Potomac, Middle Potomac, and Patuxent basins had mean TN values between 1-2 mg/l. The remaining five Tributary Strategy Basins, the Upper Western Shore, Choptank, Upper Eastern Shore, Lower Eastern Shore and Patapsco/Back River, all had TN concentrations over 2.0 mg/l. In particular, the Choptank (> 4.0 mg/l) and the Upper Eastern Shore (> 3.0 mg/l) basins were high in TN.

A similar pattern was observed for TP (Figure 14-27), with three distinct groups of mean TP values. The first group of five basins had TP levels below 0.040 mg/l (close to two ecoregion values of 0.037 and 0.031 mg/l) – this basin assemblage includes the Upper Western Shore, Lower Potomac, Upper Potomac, Patapsco/Back River and Middle Potomac. A second grouping of four basins had TP levels between 0.04 and 0.08 mg/l, and comprised the Choptank, Lower Eastern Shore, Patuxent and Lower Western Shore basins. Finally, the Upper Eastern Shore basin had a mean TP level over 0.08 mg/l – an exceptionally high TP level and of concern for Maryland nutrient strategies.

14.3.4 CORE/Trends Comparison

Fixed-site monitoring provides the best evidence of the link between nutrients in streams and receiving estuaries such as the Chesapeake Bay. The CORE/Trend program is a pre-selected, fixed station network sampled monthly to track statewide trends in water quality over time. This sampling program, started in 1974, visits 54 land-based and 3 boat sites (Sandy Point, Turkey Point -Elk River, and Kent Narrows) monthly in 12 major (6-digit) basins, year-round. Only the 54 land-based freshwater stations were considered. These fixed sites are located in 10 of the major basins, 37 of the 8-digit basins sampled by the MBSS. MBSS chemistry samples were collected during the Spring Index Period March and April, from 2000 through 2004 in all basins. Only the 560 regular and sentinel samples collected by MBSS common to the CORE/Trend were included in the comparison. Similarly, 528 CORE/Trend samples collected only during the same time period as covered by the MBSS sampling (March and April, 2000 through 2004) were considered. Chemistry analytes collected by both programs were nitrate (NO_3), total nitrogen (TN), ammonia (NH_3 as N, MBSS)/ammonium (NH_4 as N, CORE/Trend), orthophos-phate (PO_4), and total phosphorous (TP). Conductivity

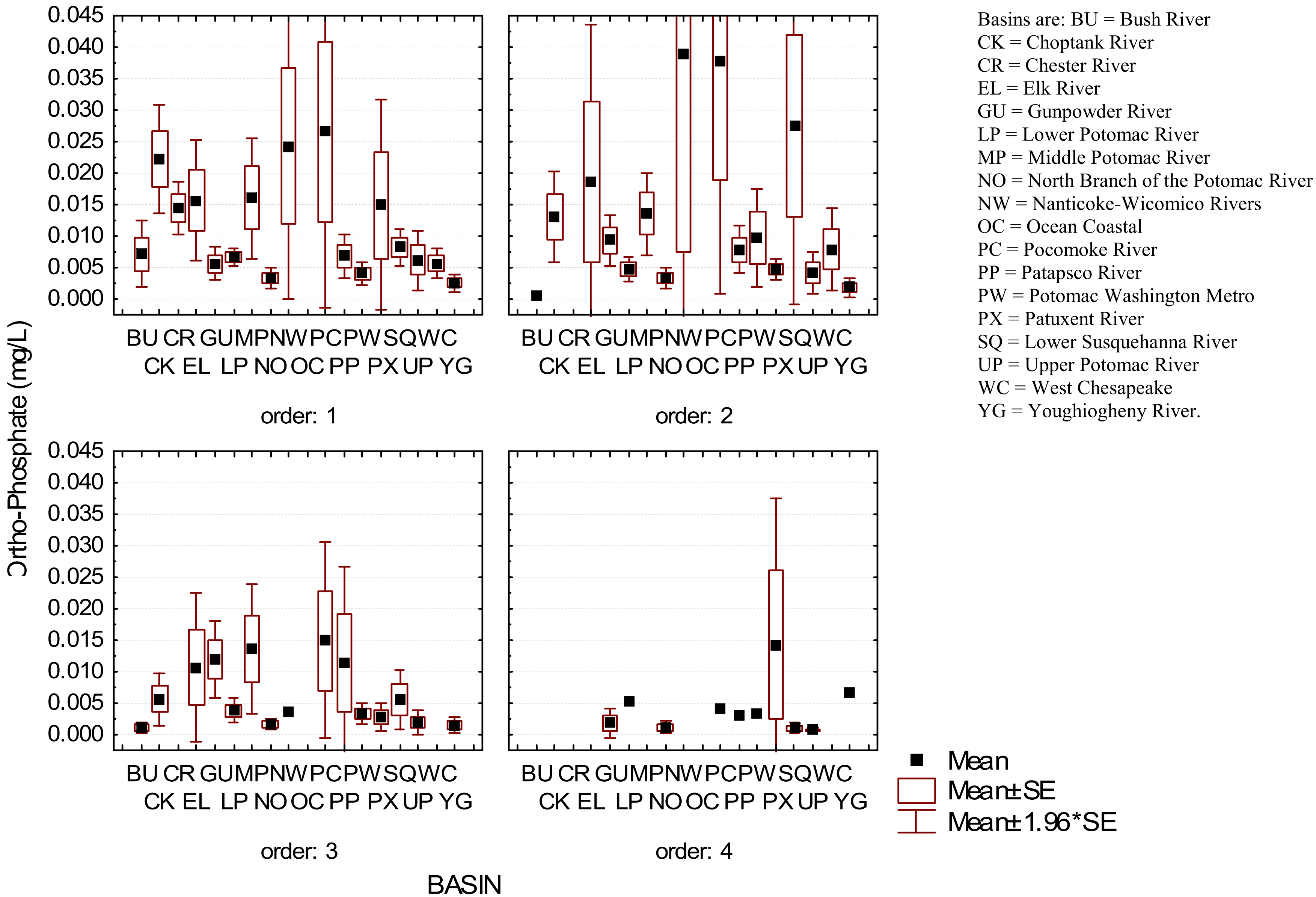


Figure 14-23. Ortho-phosphate (OP - mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.

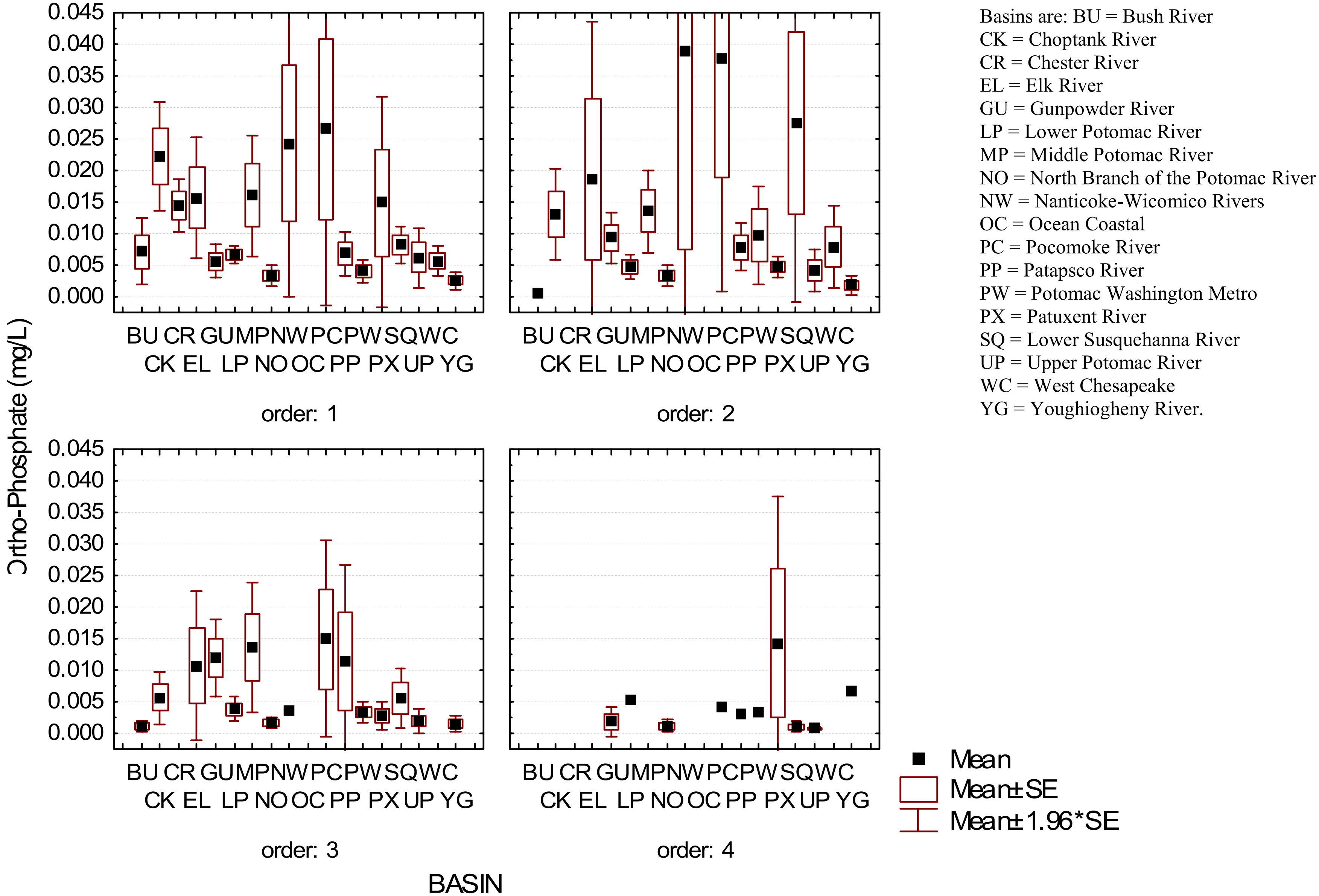


Figure 14-24. Dissolved oxygen (DO - mg/l) concentration for all basins by stream order (1-4) for 1995-2004 MBSS.

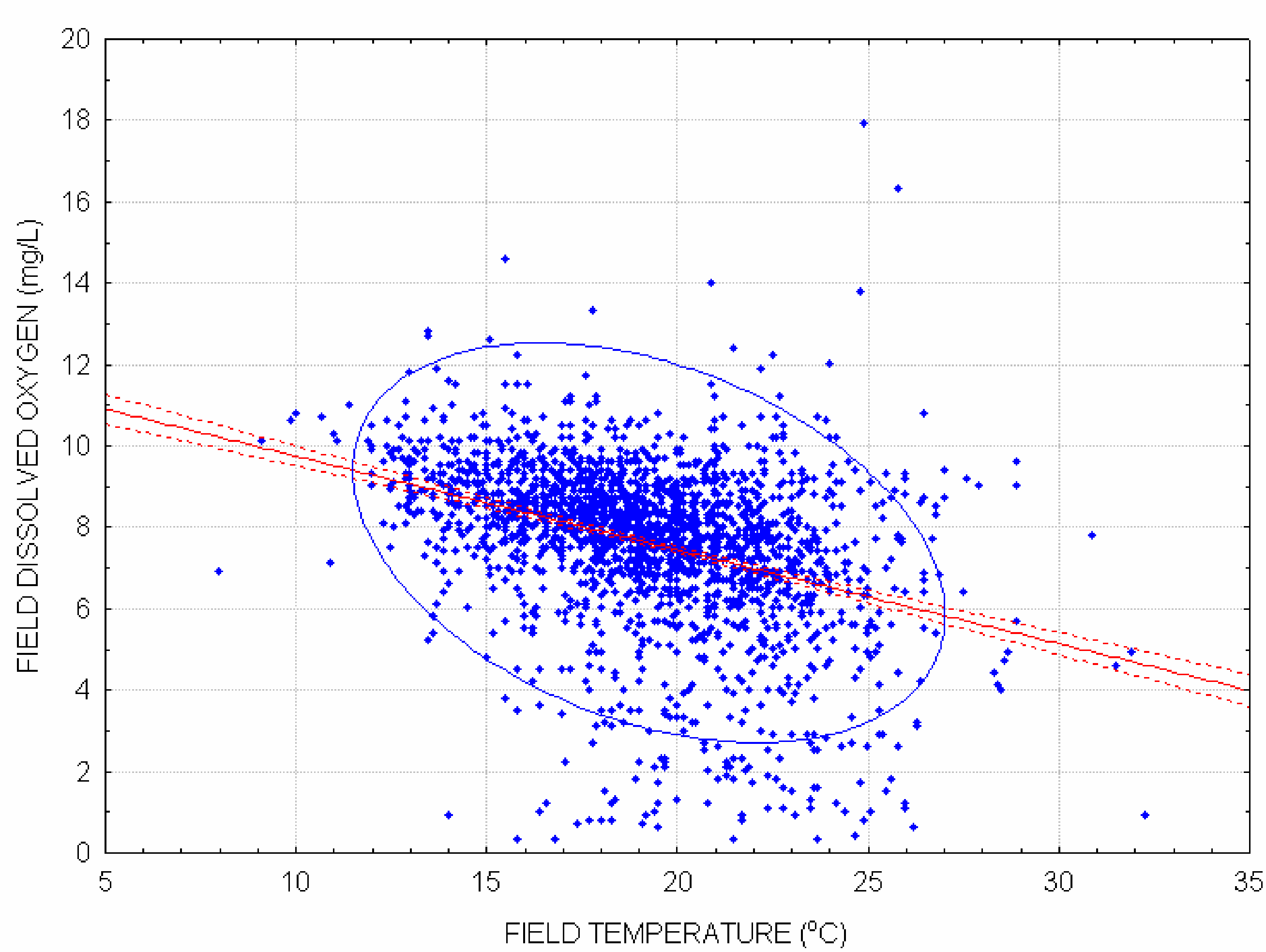


Figure 14-25. Dissolved oxygen versus field temperature for all basins by stream order (1-4) for 1995-2004 MBSS. Regression, 95% confidence interval, and 95% ellipse are shown.

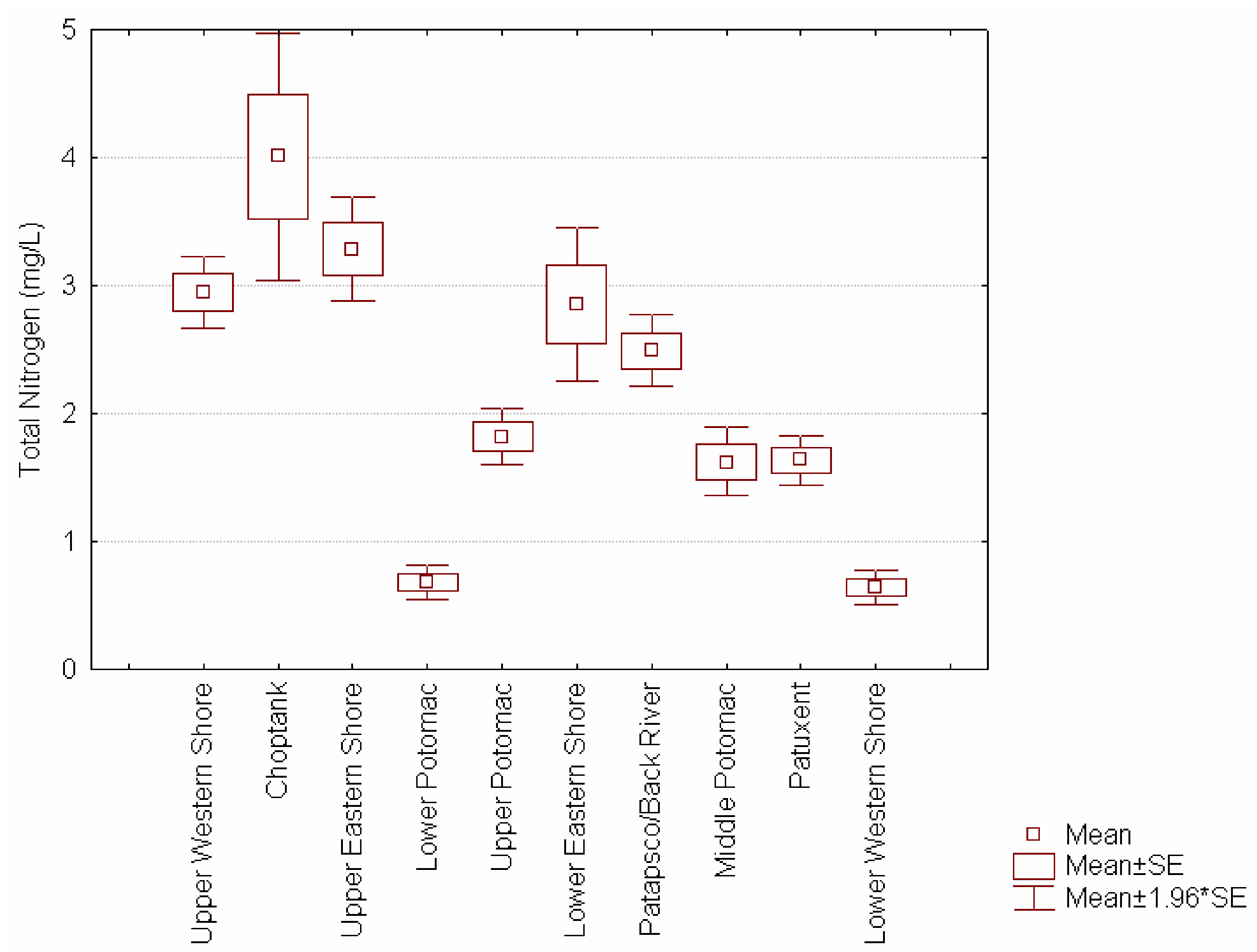


Figure 14-26. Total nitrogen (mg/l) concentration for all tributary strategy basins.

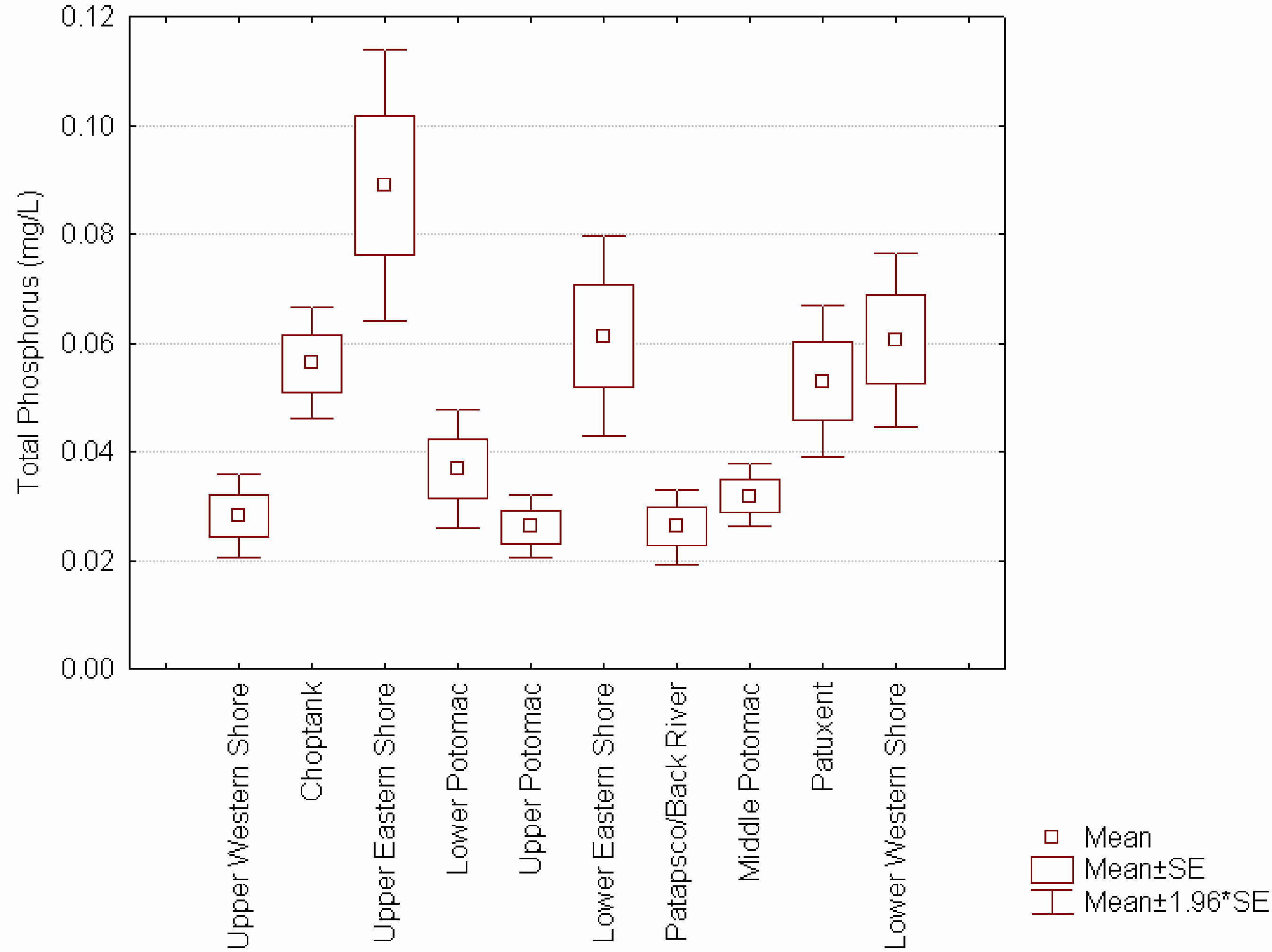


Figure 14-27. Total phosphorus (mg/l) concentration for all tributary strategy basins.

and pH were measured in-situ by the CORE/Trend program, but were lab-generated numbers for the Spring MBSS samples.

Three analyses were run: (1) included all common watersheds, (2) excluded watersheds with tidal sites, and (3) excluded sites with > 4^{th} order streams. Rank correlations (Spearman correlation) were performed for each analyte to determine correlations between programs for each group of watersheds (Table 14-10). For all the common watersheds, TN showed the best correlation, possibly related to the oxidized forms of nitrogen (NO_x). Ammonia/ammonium had the lowest correlation values, probably due to the instability of this analyte. In the second analysis, the elimination of tidal-fresh CORE/Trend basins from the analysis did not appreciably improve the correlations and the NHx correlation decreased with the elimination of tidal areas. The third analysis allowed an examination of the effect of the stream order. All correlations based on the remaining basins increased except orthophosphate. An artifact of rank correlations is the "perfect score" shown in the pH correlation. The values were different, yet the *ranks* were the same.

Changes in correlations may be in part because CORE/Trend sites are targeted to capture specific areas or conditions, such as being downstream from major point-source discharges. There are also differences in sample processing in both the field and the laboratory that could make a difference. No laboratory QC check common to both processing labs was noted. Because the CORE/Trend program uses the same sampling points, there is less variation in these data than the corresponding MBSS data. Simple summary statistics also point out differences in the detection limits and data handling decisions that can skew comparisons in areas where a particular analyte detection value is higher. For example, the MBSS TP detection limit is 0.001 and the CORE/Trend detection limit is 0.01. Summary statistics for the MBSS indicate, from the median and modes, that many of the MBSS sample values are below the CORE/Trend detection limit. Overall, there is fairly good statewide agreement between the sampling programs based on the basins they have in common when analyzed at the area scales both programs were designed to cover. Stream order appears to have an effect more pronounced in the nutrient species than in the total nutrient analytes. This is expected, as nutrient cycling and dilution change with the size of the stream. CORE/Trend values are frequently higher than MBSS values, possibly due to a combination of the effects of stream order, lab detection limits, and the fact that CORE/Trend sites are deliberately chosen to monitor specific conditions. The large discrepancy in the ammonium values (Figure 14-28) in Conewago/Double Pipe Creek (CODP) values is the result of one sample where the ammonia ion constituted most of the total nitrogen of the sample. Examination of the records showed no problems in the field or laboratory, and 3/4" of rain during the two days previous to sampling. Total nitrogen was about 10 times the total phosphorous value, and all nutrients measured were substantially above other sites on this creek. The data are consistent with storm surge runoff traveling downstream.

14.3.5 Nutrients and Biological Characteristics

Nutrient relationships with stream biotic components, and their derived indices, are often difficult to isolate from complex data sets such as the MBSS, where multiple stressors may be working at the reach to landscape level. However, there are two examples-one of the relationship between the TN/TP ratio and number of EPT taxa and the other between total phosphorus and number of EPT taxa-that illustrate nutrient effects on stream biota (Figures 14-29 and 14-30). As the TN/TP ratio increases, there is a general decrease in the number of EPT taxa present in the stream sample (Figure 14-29). Benthic macroinvertebrates, and other stream biota, have this general pattern of decreasing taxa richness with increasing nutrient loading. Ephemeroptera, Plecoptera, and Trichoptera (EPT), all taxa generally sensitive to stream degradation, are excellent indicators of nutrient pollution, and are used extensively in assessing water quality. Another nutrient-biotic example is the relationship of the number of EPT taxa with total phosphorus (Figure 14-30). EPT richness decreases as TP increases, with the maximum number of EPT observed at TP concentrations less than 0.05 mg/l–very close to the ecoregion TP thresholds. The number of EPT taxa is normally less than four at TP levels greater than 0.1 mg/l.

Table 14-10. Summary of Spearman Rank Correlations for different groups of watersheds.

Correlation Summary:	NO_3	NHx	TN	PO_4	TP	pH	COND	MBSS Sample Count	CORE/ Trend Sample Count
All watersheds	0.622	0.045	0.784	0.545	0.480	0.714	0.541	560	528
Non-tidal watersheds	0.608	0.018	0.799	0.585	0.492	0.732	0.519	535	508
No watersheds with > 4^{th} order sites	0.706	0.084	0.891	0.511	0.608	1.000	0.647	292	244

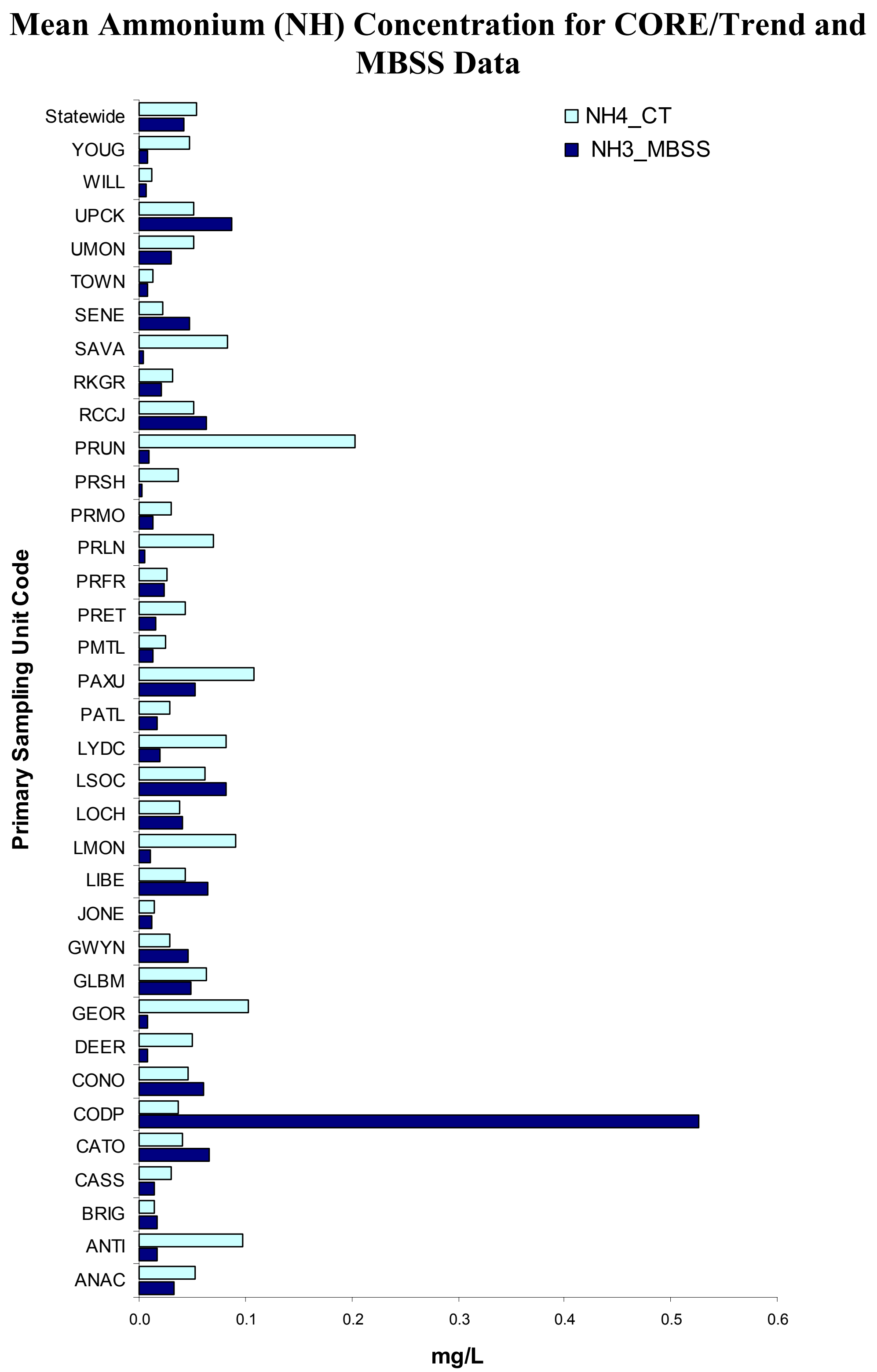


Figure 14-28. Mean ammonium concentration for CORE/Trend (NH_4) and MBSS (NH_3) stations sampled in March and April 2000 through 2004.

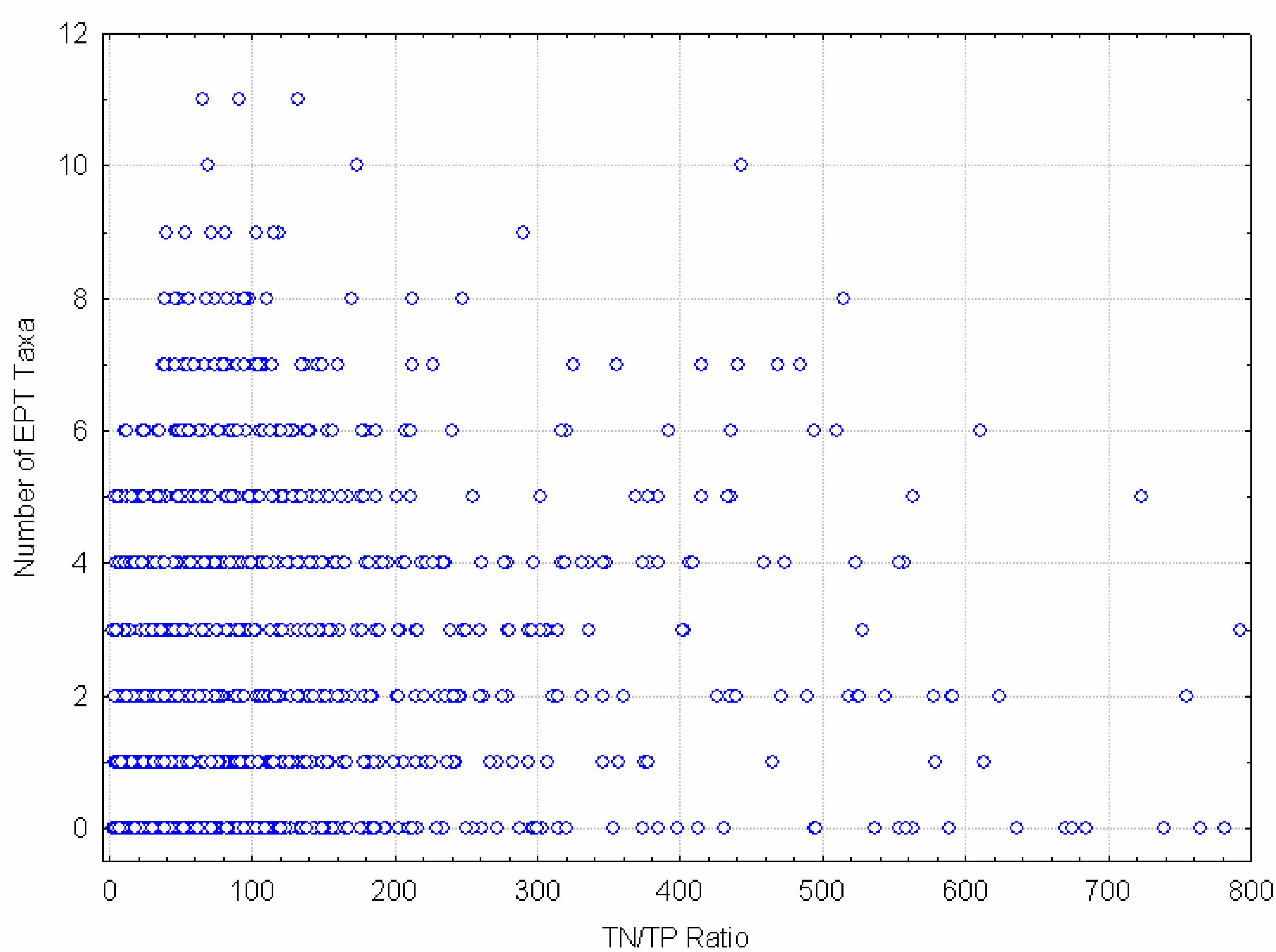


Figure 14-29. Number of EPT taxa as a function of the TN/TP ratio, 2000-2004 MBSS.

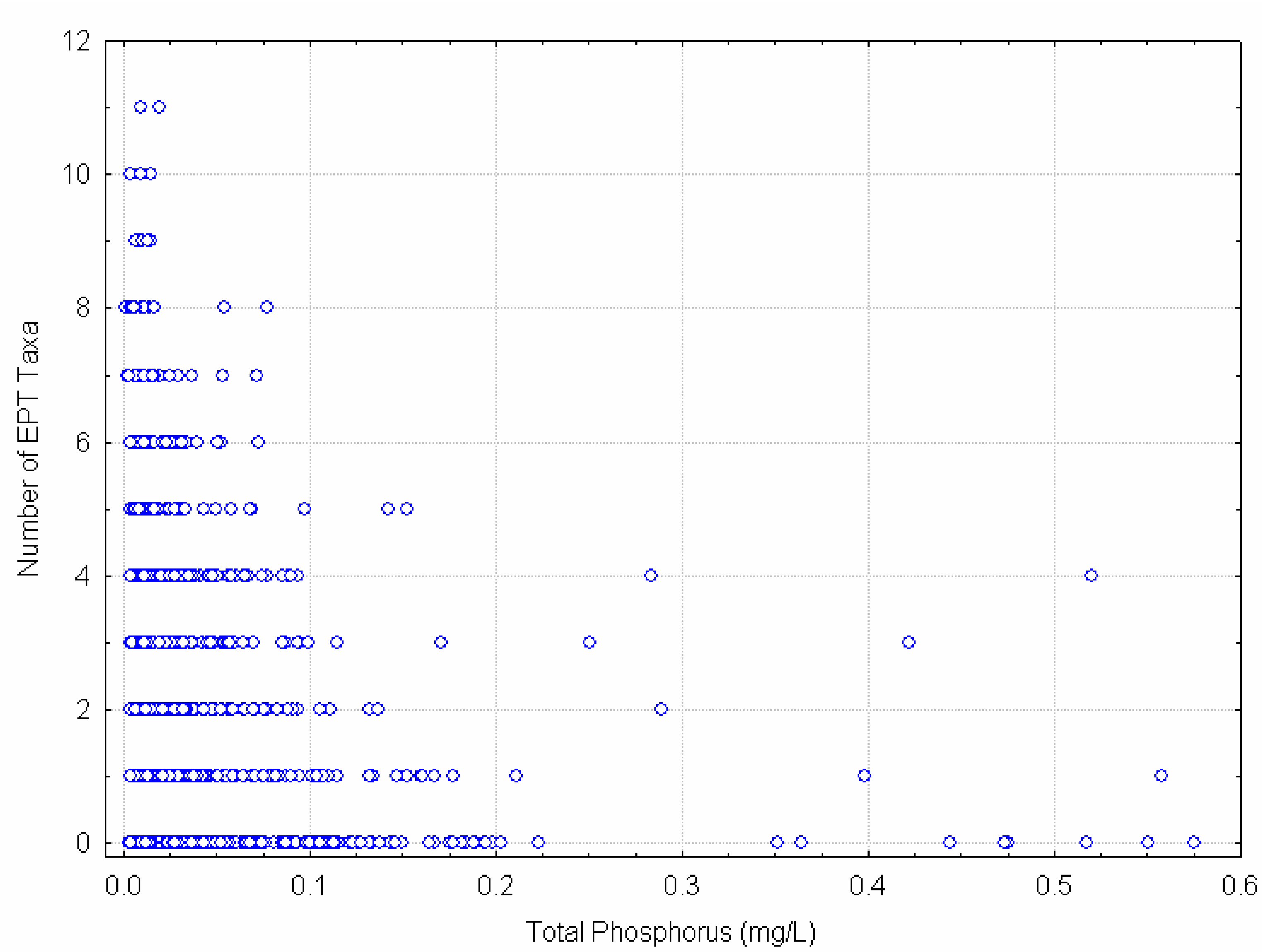


Figure 14-30. Number of EPT taxa as a function of stream total phosphorus (TP), 2000-2004 MBSS.

14.3.6 Storm Nutrient Dynamics

The MBSS samples water quality during the spring index period, corresponding to the spring baseflow for Maryland streams, with some additional field water quality taken during the summer index period sampling. Although the spring index sample is a point water quality sample taken during the year, the MBSS does represent a comprehensive spatial picture of first- through fourth-order non-tidal streams in Maryland. Of paramount interest to Maryland is stream nutrient dynamics, especially given the linkage between headwater systems (first- and second-order) and the tidal reaches of the Chesapeake Bay. In this regard, stream nutrient dynamics are important to consider in managing fluxes of organic and inorganic materials (Peterson et al. 2001, Gomi et al. 2002); including dynamics of riparian buffers in mitigating nutrient inputs (Sweeney et al. 2004). Small streams (comprising 76% of Maryland stream km) are responsible for the most rapid uptake and transformation of inorganic nitrogen (Peterson et al. 2001). Nutrient reduction is a goal of the Maryland Tributary Strategy, and this goal reflects a basin approach to cleaning up the Chesapeake Bay.

Both TN and TP (Figures 14-26 and 14-27) figure prominently in strategies to reduce nutrient loadings (U.S. EPA 2000a,b,c, Dodds and Welch 2000, Pinay et al. 2002, King and Richardson 2003), especially since the U.S. EPA is responsible for setting nutrient criteria for streams, rivers and lakes as part of the Clean Water Action Plan. TN and TP are relatively easy to measure, even at low levels, and eliminate certain analytical problems in measuring some nitrogen species (e.g. DIN) found in water. There is also a significant body of limnological literature that examines TN:TP ratios – known throughout the literature as the Redfield ratio, where C: N: P is found in the ratio of 106:16:1 in algal tissue, and serves as a model to examine nutrient limitations (Allan 1995, Dodds 2002). This concept of a 16:1 TN:TP ratio has been discussed extensively and stands as a nutrient paradigm, although there are questions as to its utility in lotic systems (Allan 1995, Dodds 2002). The classic work by Omernik (1977), done for small and relatively unpolluted streams throughout the U. S., serves as a benchmark to assess TN:TP ratios, where ratios of ~12:1 are associated with > 90% forest, 21:1 with > 50% forest, 26:1 with > 50-75% agriculture, and 60:1 with agriculture > 90%.

Maryland TN:TP ratios, as plotted for all tributary strategy basins and the Youghiogheny and Ocean Coastal basins, reveal some interesting results (Figure 14-31). Mean TN:TP ratios for the Lower Potomac, Ocean Coastal, and Lower Western Shore are all below a TN:TP ration of 40, with four basins ranging from a TN:TP ratio of 60-100 and five having ratios greater than 100. Since the average TP for the 2000-2004 MBSS is 0.041 mg/l (median = 0.020 mg/l), and the 75% percentile value is 0.043 mg/l TP, these TN:TP ratios > 60 indicate that there are basins leaking nitrogen into tidal reaches of the Chesapeake Bay, where nitrogen is a limiting nutrient for phytoplankton growth in mesohaline and coastal marine waters (Paerl et al. 1990). It appears that TP is being highly conserved, but TN is being leaked in MBSS streams. Using the U. S. EPA Aggregate Ecoregion approach for rivers and streams, the TN:TP ratio for Ecoregion IX is 19, for XI 31, and for XIV 23 – all MBSS basins are above these ecoregion threshold ratios.

To better illustrate nitrogen (TN) and phosphorus (TP) levels in Maryland, these two analytes were plotted, by percentiles, for all MBSS sites. There were a number of TN values that exceeded the 90th percentile on the Eastern Shore, and throughout Central Maryland. In concordance with these high TN, sites exceeding the 50-75th and 75-90th TN percentiles were common in Central Maryland. There were numerous MBSS sites in the TP 75-90th percentile, clustered on the upper Eastern Shore and Choptank, the lower Potomac River, the Lower Patuxent River and the Upper Potomac River. Few TP values exceeded the 90th percentile. It is obvious that nutrient reduction strategies should focus on these regions of Maryland.

14.4 PHYSICAL HABITAT

While acidification, nutrient enrichment, and other water pollution are well known causes of degradation, MBSS data consistently show that physical habitat changes are the primary cause of biotic impoverishment in Maryland streams (affecting more than 50% of stream miles based on Roth et al. 1999 and Volume 7 of this report). The MBSS Physical Habitat Index (PHI, Paul et al. 2003) is a measure of how much physical habitat in streams varies from reference condition; PHI scores by Tributary Basin and County are reported in Volumes 7 and 8 of this report, respectively. Individual metrics collected by the MBSS provide evidence of the specific physical habitat stressors and probable causes of degradation, and are the focus of this section.

Three candidate causes affect physical habitat in streams: temperature, channel alteration, and sediment. Temperature changes primarily result from (1) removal of canopy cover and direct heating of the stream and (2) heated runoff from impervious surfaces. Channel alteration includes direct channelization (often including armoring), creation of impoundments or fish barriers, and changes in fluvial geomorphology that result from altered flows. The input of terrestrial sediments results from changes in land use, especially in the riparian zone. Sediment impacts on the availability of instream habitat can also result from flow regime changes (higher or “flashier” flows) that erode stream banks. Separating the ultimate causes of sedimentation can be problematic. The role of riparian

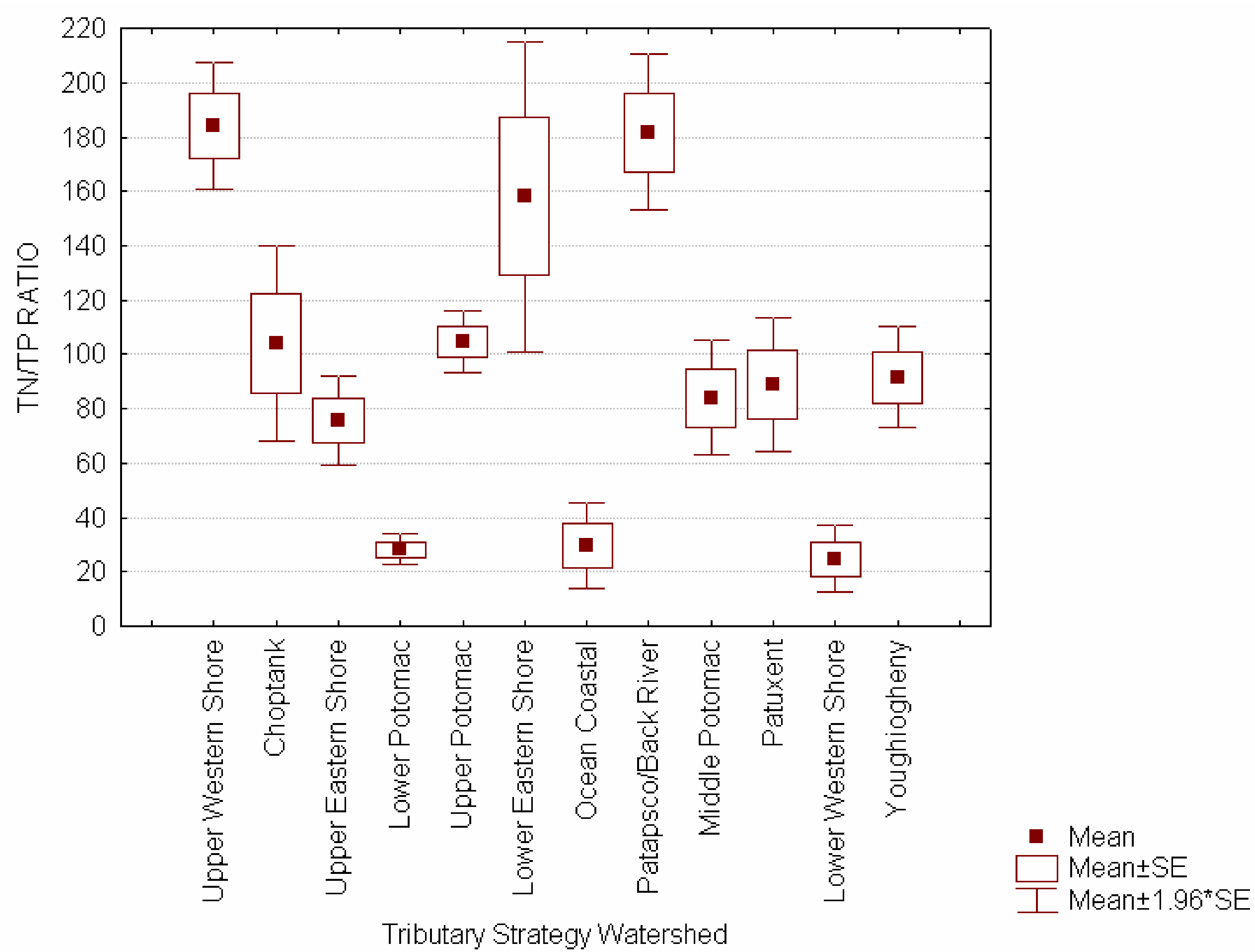


Figure 14-31. TN/TP ratios for all tributary strategy (Youghiogheny and Ocean Coastal) basins.

(streamside) vegetation in modifying stressors to streams is so important that it is addressed in detail in Volume 10.

14.4.1 Background

Stream health, as determined by the condition of biological communities, has been shown to be directly correlated to physical habitat quality (Richards et al. 1993, Rankin 1995, Roth et al. 1999). Habitat loss and degradation has been identified as one of six critical factors affecting biological diversity in streams worldwide; habitat alteration is cited as a leading cause of fish species extinctions, contributing to 73% of extinctions in North America during this century (Miller et al. 1989, Allan and Flecker 1993). Habitat degradation can result from a variety of human impacts occurring within the stream itself or in the surrounding basin. Urban development, agriculture, timber harvesting, livestock grazing, and the draining or filling of wetlands are the best known human activities affecting streams.

Alone or in combination, these human activities may cause changes in vegetative cover, sediment loads, hydrology, and other factors influencing stream habitat quality. The amount of vegetative cover in a basin regulates the flow of water, nutrients, and sediments to adjacent streams. In basins under anthropogenic stress, riparian forests can ameliorate inputs of nutrients, sediments, and other pollutants to streams. They also provide local benefits of shade, overhead cover, leaf litter to feed the aquatic food web (allochthonous input), and large woody debris, which in turn provides cover and forms pool and riffle microhabitats (Karr and Schlosser 1978, Gregory et al. 1991). Removal of riparian vegetation can increase stream temperatures, often with adverse effects on stream fish (Barton et al. 1985). The loss of basin or riparian vegetation increases the potential for overland and channel erosion, often increasing the siltation of stream bottoms and obliterating the clean gravel surfaces used by many fish species as spawning habitat (Berkman and Rabeni 1987). Stream bottoms that become embedded with increased sediment loads provide less habitat for many benthic macroinvertebrates. Stream channelization alters runoff patterns and creates "flashy" streams with more extreme high and low flows, increased scouring and streambank erosion. These altered flows accelerate downcutting and widening of stream channels. This increased hydrologic variability is exacerbated by urbanization, which increases the amount of impervious surface in a basin and causes higher overland flows to streams, especially during storm events. Streams with highly altered flow regimes often become wide, shallow, and homogeneous, resulting in poor habitat for many fish species (Schlosser 1991). Concrete-lined streams are perhaps the most severe example of habitat loss for fish, benthic macroinvertebrates, and other aquatic species.

14.4.2 Physical Habitat Index

The MBSS Physical Habitat Index (PHI) was developed to describe the physical habitat component of freshwater streams that strongly influences the composition and status of stream fish communities (Gorman and Karr 1978). As described in Volume 6: Laboratory, Field, and Analytical Methods, MBSS procedures for physical habitat assessment were derived from two sources: EPA's Rapid Bioassessment Protocols (RBPs) (Plafkin et al. 1989) as modified by Barbour and Stribling (1991), and the Ohio EPA's Qualitative Habitat Evaluation Index (Rankin 1989). In addition to the 13 qualitative physical habitat metrics derived from these methods, qualitative and quantitative stream characteristics (meandering, presence of emergent and submerged vegetation, presence of coarse woody debris, rootwad number, etc.) were recorded during MBSS field sampling. All of the measured parameters were considered in the development of a reference-based PHI for Maryland streams.

The revised PHI was developed using MBSS data through 2000 (Paul et al. 2003). Because of underlying differences in stream types, separate PHIs are developed for each of three geographic strata: the Highlands, Piedmont, and Coastal Plain. Four physical habitat variables are common to all three indices: (1) bank stability, (2) epifaunal substrate, (3) shading, and (4) remoteness. Five additional variables are included in one of two indices: (1) riparian buffer width, (2) riffle quality, (3) instream wood, (4) instream habitat quality, and (5) embeddedness.

Index scores were adjusted to a centile scale that rates each sample segment as follows:

- Scores of 81 to 100 are rated minimally degraded
- Scores of 66 to 80 are rated partially degraded
- Scores of 51 to 65 are degraded
- Scores of 0 to 50 are rated severely degraded

Figure 14-32 shows the PHI score for the 1,065 MBSS sites sampled from 2000-2004. The percentage of stream miles with each PHI rating are provided by Primary Sampling Unit (PSU) generally Maryland 8-digit watersheds) in Volumes 1 through 5, and for Tributary Basins and the entire State in Volume 6.

Stream mile estimates of PHI scores indicate that approximately one-third (33%) of Maryland streams have degraded to severely degraded physical habitat. Only 21% of streams have minimally degraded physical habitat. The extent of degraded physical habitat throughout Maryland is likely a result of several factors. The following sections discuss each of these factors and their candidate causes in turn.

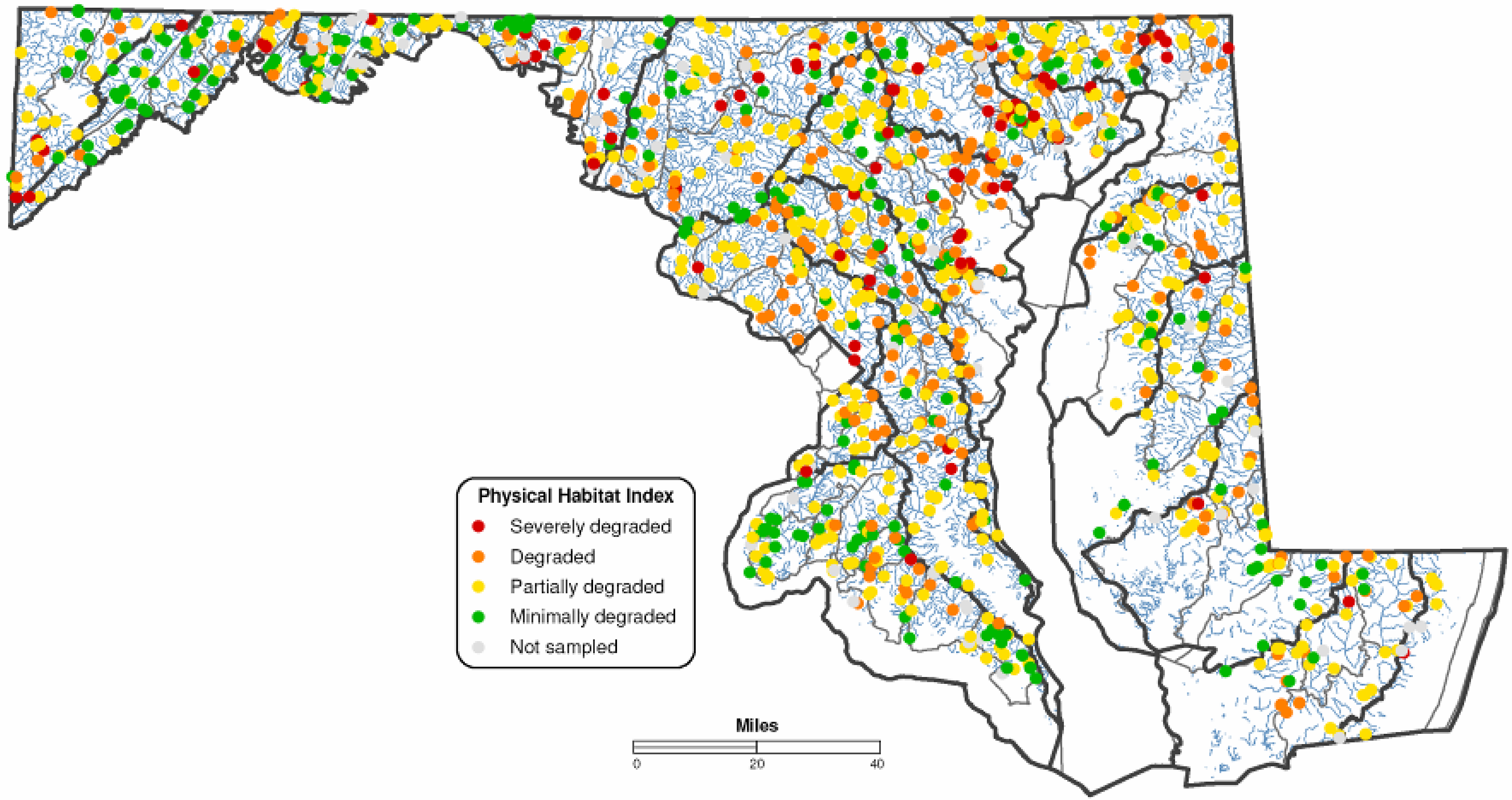


Figure 14-32. Geographic distribution of Physical Habitat Index (PHI) ratings for sites sampled in the 2000-2004 MBSS. Ratings are as follows: 81-100 good, 66-80 fair, 51-65 poor, and 0-50 very poor.

14.4.3 Riparian Buffer

The width and quality of riparian buffer strongly alters the amount of runoff into streams and the resultant stress from temperature, erosion, sediment, and flow regime. In addition, the presence of riparian vegetation affects the amount of woody debris and other allochthonous input (leaf fall) that affect stream structure and energy processing. Volume 10 discusses this topic in detail, so only a brief summary is provided below.

MBSS results describe both the type and extent of local riparian vegetation, estimated as the width of the riparian buffer along each 75-m sample segment. Statewide, an estimated 65% of stream miles had forested buffers on both sides of the stream, 12% had other kinds of vegetated buffers (wetland, old field, tall grass, or lawn) on both sides, and 10% had no buffer (sites were not included in the analysis if an outfall pipe was observed draining directly into the stream segment or if severe buffer breaks were present). An estimated 58% of stream miles had at least a 50-m riparian buffer on both sides of the stream (Figure 14-33). The data indicate that as buffer width increases, buffer type switches from roughly an even split between forest and other vegetation to nearly entirely forested buffer.

A statewide map (Figure 14-34) shows the distribution of riparian buffer widths observed at MBSS sites. Sites with at least a 50-m vegetated buffer were distributed throughout the state. The largest concentrations of sites with no buffer or buffer widths of less than 50-m were in the Lower Easter Shore basin and portions of the Baltimore-Washington corridor; other sites with less than a 50-m buffer were scattered throughout the state.

Estimates of the extent of stream miles lacking riparian buffer indicated that 10% of stream miles statewide had no buffer, while another 5% had only a vegetated buffer 1-5 m wide on the side with the least amount of buffer. The Lower Eastern Shore and Youghiogheny basins had the largest percentage of poorly buffered stream miles, with 18% lacking any buffer and 4% with 1-5 m of vegetation for each basin. Fifteen percent of stream miles in the Upper Potomac basin were unbuffered, while another 8% had 1-5 m of vegetation for the least buffered side of the stream. For all basins buffer width less than 5 meters on the least buffered side ranges between 6% (Lower Western Shore) to 25% (Lower Eastern Shore). The problem of insufficient riparian buffer is widespread throughout the State, presenting numerous opportunities for stream restoration through re-establishment of trees and other vegetation along riparian corridors. Riparian restoration efforts should be targeted to areas with the greatest potential for ecological benefit (e.g., reduced nutrient runoff, enhanced stream habitat, and improved water quality).

14.4.4 Temperature

Streams are adapted to their natural temperature regimes. Natural stream temperatures depend on climatic region, elevation, groundwater inputs, and riparian vegetation. Regular groundwater inputs and shading from riparian trees create cooler temperatures in any region. As Maryland's forests have been replaced with other land uses, riparian shading is frequently lost and groundwater inputs decrease as infiltration is reduced by less pervious surfaces. The water directly entering streams itself is warmed by the impervious surfaces and pipes that drain most urban areas (Walsh et al. 2004). The more open channels and shallower depths of urban streams also likely contribute to greater variation in stream temperatures between day and night. Warmer water can worsen the problems of algal growth, thereby affecting the natural energy processing in streams. Unnaturally warm water (thermal pollution) can also arise from small farm dams (Lessert and Hayes 2003) and constructed stormwater treatment ponds (Walsh et al. 2004).

Since 1997, the MBSS has deployed continuous reading temperature loggers at more than 2,000 sites. The long-term goal is to use temperature data to (1) better classify and characterize coldwater streams and (2) identify streams stressed by temperature changes, such as spikes from rapid inputs of warm water running off impervious surfaces during summer storms. Data were recorded at 20-minute intervals with loggers set to record the highest value observed during each 20-minute interval. Because temperature loggers are sometimes lost or not submerged in the stream during low flow periods, careful examination is needed to establish a consistent period of record and compute meaningful summary indicators such as:

- Mean average daily temperature
- Mean minimum and maximum daily temperatures
- Absolute maximum temperature
- 95th percentile temperature
- Percentage of readings exceeding thresholds in state water quality standards

Ultimately, the MBSS plans to analyze all MBSS temperature data and compare it to Maryland water quality standards for temperature, which state that the maximum temperature may not exceed 32 EC (90 °F) in most waters, 20 °C (68 °F) in Class III Natural Trout Waters, or 23.9 °C (75 °F) in Class IV Recreational Trout Waters (COMAR 1995). EPA criteria for growth and survival of brook trout (Maryland's only native salmonid) are maximum weekly means of 19 and 24 °C. Research has found that a still lower temperature of 14.4 °C is the maximum temperature for juvenile growth of brook trout (EPA 1976 and McCormick et al. 1972, as cited in Eaton 1995).

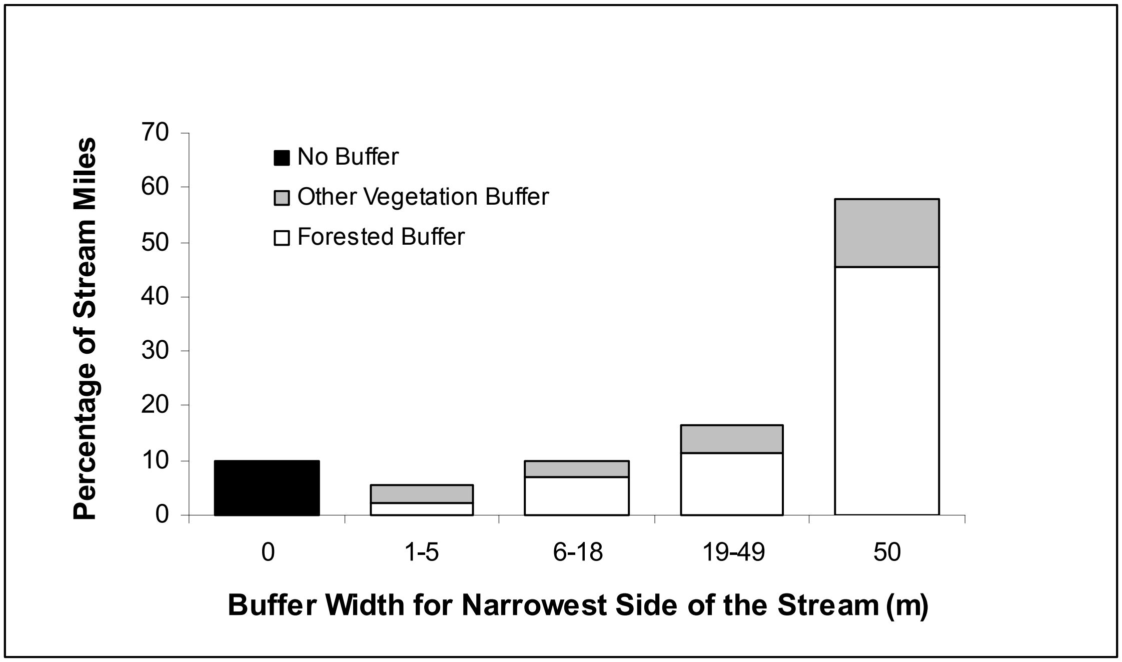


Figure 14-33. Percentage of stream miles by riparian buffer type and width for the 2000-2004 MBSS. The category "Other Vegetation Buffer" includes old field, emergent vegetation, mowed lawn, tall grass, and wetland vegetation. No (effective) buffer indicates that although some vegetation may be present, runoff (such as from an outfall pipe) occurs directly into the stream.

Figures 14-35 and 14-36 show the temperature records of typical warmwater and coldwater sites. Note that in both cases a significant proportion of days between June and September exceeded the respective temperature thresholds.

14.4.5 Channel Alteration

Dredging, filling, and construction in a stream channel are the most direct ways to affect physical habitat. Channelization refers to both the straightening of channels and their armoring with concrete or other hard materials. Dams alter upstream areas by converting lotic stream habitat to lentic (ponded) habitat, resulting in silt deposition, increased water temperature, and barriers to the movement of fish. Even small culverts or other structures (exposed sewer pipes) in the stream channel can block fish movement. In addition, beaver dams can flood large areas, dramatically changing stream character.

14.4.5.1 Stream Bockages as Stressors to Stream Communities in Maryland

Stream blockages such as dams, weirs, and culverts can prevent migratory fishes access to upstream habitats and have been responsible for the reduction or elimination of populations of migratory species throughout the world, including Chesapeake Bay. Reduction or the complete loss of populations of anadromous species (e.g., American shad, hickory shad, alewife herring, blueback herring, white perch, yellow perch, striped bass) and catadromous species (e.g., American eel) from many tributaries to Chesapeake Bay as a result of stream blockages have resulted in concerted efforts to restore fish passages and re-establish populations of these commercially important species.

The Maryland DNR Fisheries Service began surveying blockages to anadromous fish passage in the late 1960s and 1970s. Surveys documented more than 1,000 fish blockages to migratory species including dams, culverts, gauging weirs, and sewer lines (Figure 14-37). In 1987, the first Chesapeake Bay Agreement was signed by states within the basin, including Maryland, Pennsylvania, Virginia, and the District of Columbia. This landmark agreement included commitments within each state "to provide for fish passage at dams, and remove stream blockages wherever necessary to restore passage for migratory fish." The Bay states agreed to reopen 1,357 miles (thought to be the majority of historic stream miles available) of historical spawning grounds by the year 2003, of which Maryland's share was approximately 389 stream miles. In response to this commitment, the Maryland Department of Natural Resources created the Fish Passage Program in 1988.

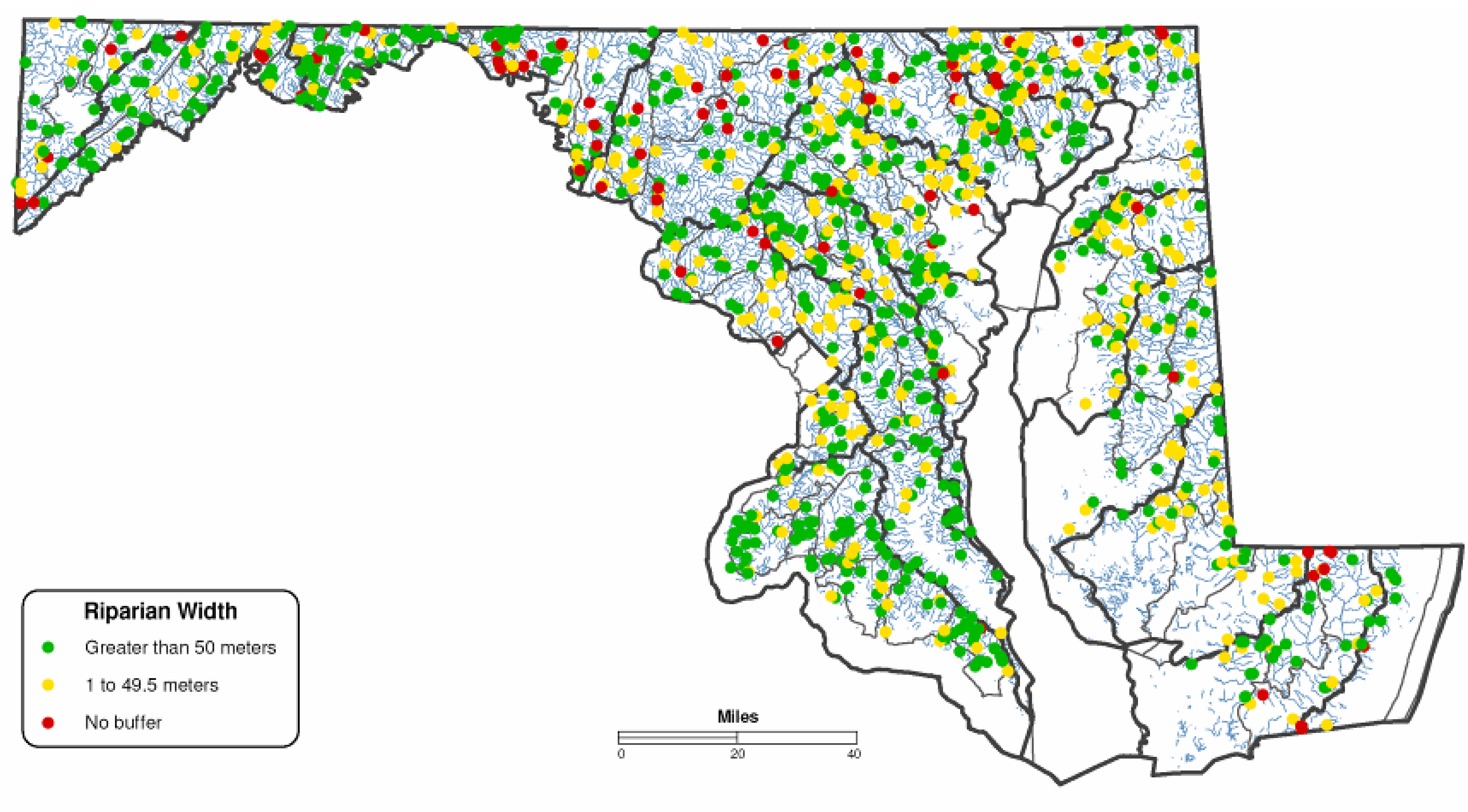


Figure 14-34. Riparian buffer width at sites sampled in the 2000-2004 MBSS.

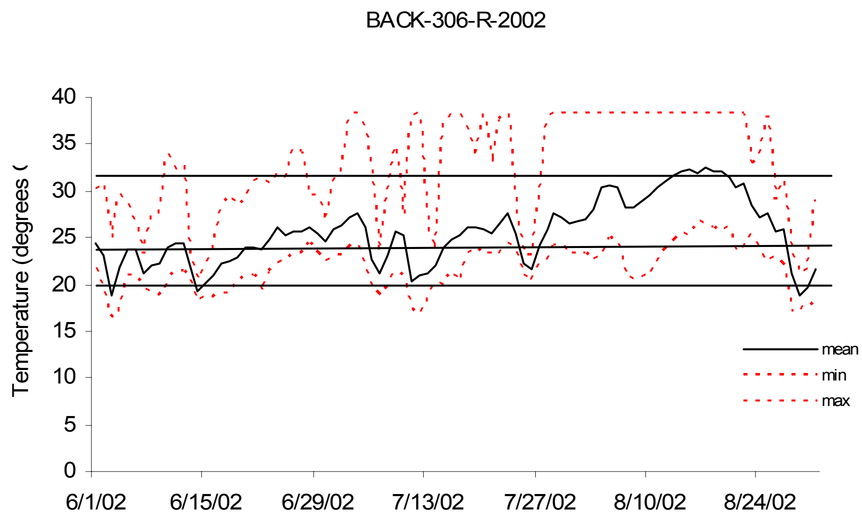


Figure 14-35. Mean, minimum and maximum daily temperatures (degrees Celsius) for a warmwater stream sampled in MBSS 2002, BACK-306-R-2002. Period of record was from June 1, 2002 to August 31, 2002. Horizontal threshold lines indicate Maryland water quality standards maximums for Class III, Class IV, and other waters.

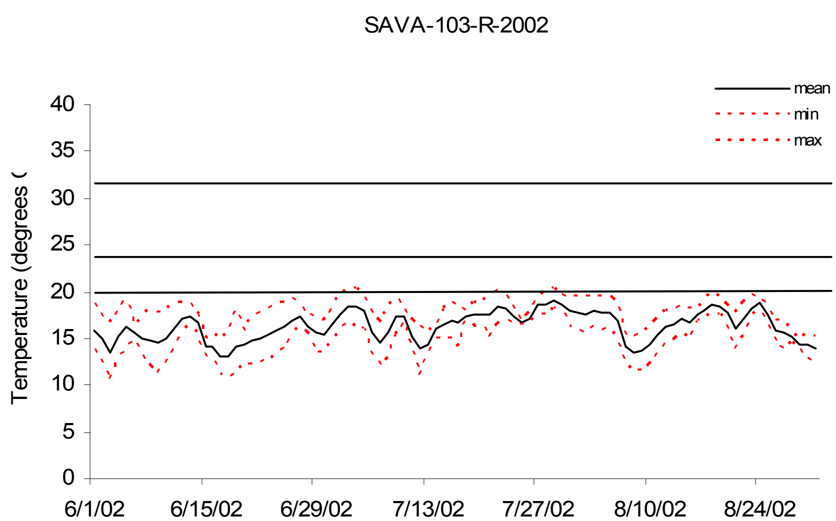


Figure 14-36. Mean, minimum and maximum daily temperatures (degrees Celsius) for a coldwater stream sampled in MBSS 2002, SAVA-103-R-2002. Period of record was from June 1, 2002 to August 31, 2002. Horizontal threshold lines indicate Maryland water quality standards maximums for Class III, Class IV, and other waters.

The Fish Passage Program's purpose is to restore migratory fish species to historic or near-historic levels of the 1950s. Since funding has been a limiting factor, the program has typically focused most of its attention on the larger blockages. With this in mind, fish passages have been provided at many of the larger blockages on Maryland waterways, including Conowingo Dam on the Susquehanna River, and Bloede, Simkins, and Daniels dams on the Patapsco River. They have also been provided at some of the smaller dams such as Fort Meade Dam on the Little Patuxent River, Van Bibber Dam on Winters Run, the dam at Elkton on Big Elk Creek, and the Tuckahoe Lake Dam in Tuckahoe State Park, as well as many others. The original goal of the Chesapeake Bay Agreement was surpassed in 2004 when the Bay states collectively opened 1,570 miles of historic spawning habitat. Installations of fish passages have reopened more than 400 miles of streams in Maryland to migratory species.

Many resident fishes are also known to move some distances to preferred seasonal habitats for spawning and feeding, and to refugia during times of stress. The

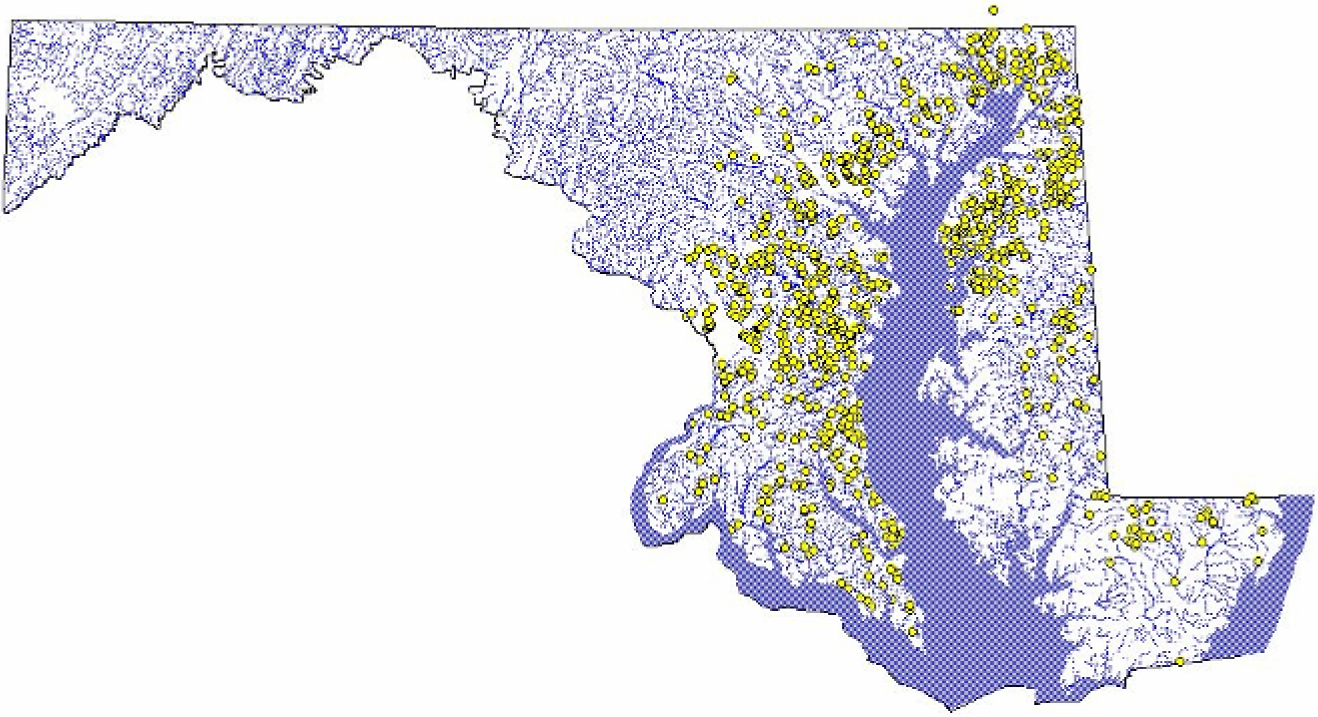


Figure 14-37. Stream blockages to the migration of anadromous fishes identified by MDNR Fisheries Service.

influence of blockages on resident fish populations and community structure can be profound. The most pervasive influence of blockages on resident fishes may be as barriers to upstream re-colonization. Blockages can interrupt pathways of immigration and emigration, and limit the exchange of individuals between populations. This change in metapopulation structure, culminating in fragmented and isolated populations upstream of a blockage, can result in local extinctions following catastrophic events (Winston et al. 1991, Dunham and Rieman 1999). These events may displace or eliminate all or part of a stream fish community, after which re-colonization is impossible. Stream blockages may more severely affect rare resident species by increasing the likelihood of local extinction (Fagan et al. 2002).

Although large stream blockages have received the most attention, small blockages such as box culverts, pipe culverts, gabion baskets, and sewer lines also are barriers to resident and migratory fish movement (e.g., Kenny et al. 1992; Gibson et al. 2005; Warren and Pardew 1998) and often degrade or reduce fish habitat (Harper and Quigley 2000). In Maryland, small blockages are numerous and widespread. For example, MBSS has identified 32 small blockages to fish passage at or near sites sampled from 1995-2004. These blockages included 16 dams, 6 pipe culverts, 4 box culverts, 3 pipeline crossings, 2 arch culverts, and 1 gauging weir (Figure 14-38). Since 1996, the MDNR Stream Corridor Assessment (SCA) has conducted stream walks within 23 Maryland 8-digit basins. In addition to documenting various aspects of stream condition, one of the goals of the SCA is to identify structures that may impede fish movement such as man-made dams, road crossings, pipe crossings, channelized stream sections, beaver dams, and natural falls (Figure 14-38). The State Highway Administration (SHA) also notes blockages to fish passage during their biannual inspection of bridges and large culverts (Figure14-38).

Poor design and/or improper installation of culverts and stream channel alterations associated with road crossings often cause complete or intermittent blockage to upstream fish movement. Culverts often result in insufficient stream depth, excluding certain species due to body size or shape. They also alter elevation of the streambed. Downstream outlets of culverts are often higher than the original streambed, creating a vertical drop that is often insurmountable. High water velocities produced by culverts may also exclude upstream movement by some species. Large cobbles, boulders, or gabion baskets placed at road crossings often create physical barriers to fish movement.

Kenny et al. (1992) surveyed 48 road crossings in Maryland during the period from 1988 to 1990. Twenty-eight (58%) were identified as being complete (15), seasonal (9), or frequent (4) barriers to fish passage. Surveys of road crossings by Gibson et al. (2005) and Langill and Zamora (2002) in Canada found that 53% and 58% served as complete or intermittent barriers to fish passage, respectively. If approximately 50% of road

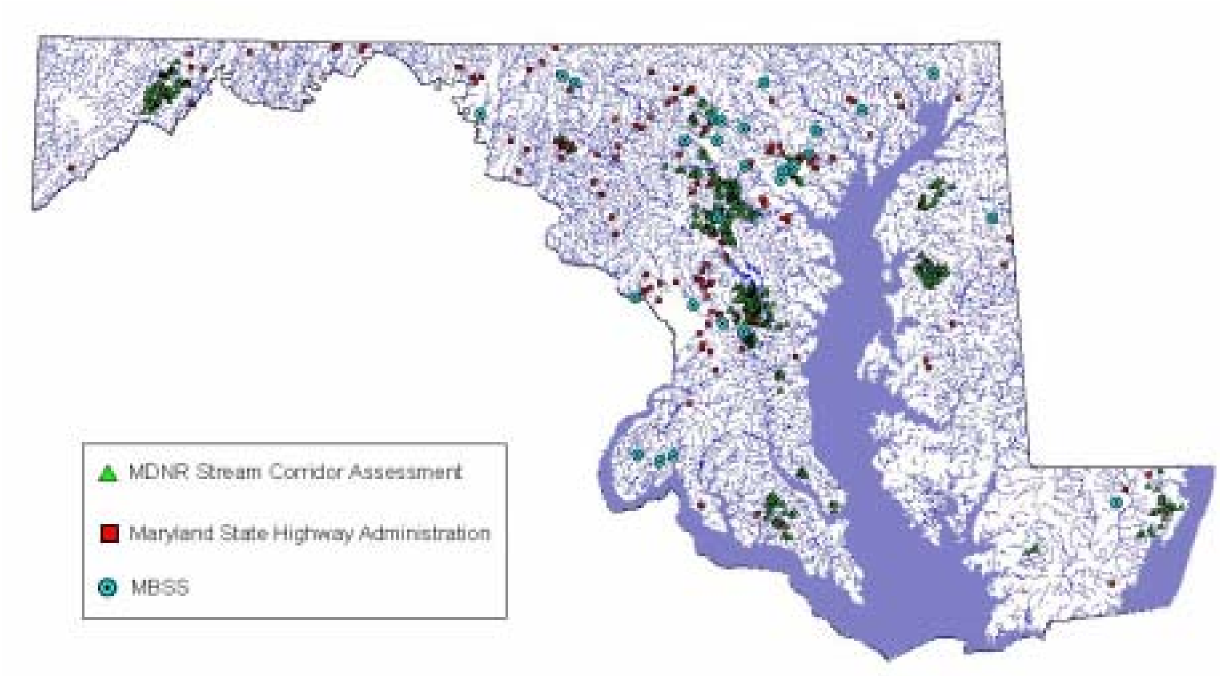


Figure 14-38. Stream blockages identified by the MBSS, MDNR Stream Corridor Assessment, and Maryland State Highway Administration.

crossings are barriers to fish migration, as found in these three studies, nearly 6,851 of an estimated 13,703 road crossings in Maryland (data from U.S. Department of Commerce Bureau of the Census Geography Division) are potentially complete or intermittent blockages.

In addition to permanent stream blockages, beaver ponds can also limit fish passage. Based on MBSS sites having beaver ponds or being unsampleable because of beaver activity, an estimated 6% of stream miles statewide had beaver ponds. The areas with the greatest extent of beaver ponds were the Lower Potomac (20%) and the Upper Eastern Shore (22%). No beaver activity was observed for Choptank, Lower Eastern Shore, and the Ocean Coastal basins. (Figure 14-39).

During 1995, fish sampling was conducted by MBSS upstream and downstream of a small blockage (1 ft. high) at a road crossing on Budd's Creek, a tributary to the Lower Potomac River basin, in Charles County, Maryland. Sampling was restricted to fish species presence or absence and equal sampling effort was used above and below the fish blockage. Eight species were collected downstream of the blockage including creek chub (*Semotilus atromaculatus*), eastern mudminnow (*Umbra pygmaea*), blacknose dace (*Rhinichthys atratulus*), tessellated darter (*Etheostoma olmstedi*), American eel (*Anguilla rostrata*), bluegill (*Lepomis macrochirus*), golden shiner (*Notemigonus crysoleucus*), and spottail shiner (*Notropis hudsonius*). Upstream of the blockage, only four of these species were collected: creek chub (*S. atromaculatus*), eastern mudminnow (*U. pygmaea*), tessellated darter (*E. olmstedi*), and golden shiner (*L. macrochirus*).

As part of a cooperative study with the U.S. Army Corps of Engineers in 1998, the MBSS sampled upstream and downstream of two concrete channels at road crossings in the Western Branch basin to determine whether the channels acted as barriers to the upstream migration of resident fishes. Despite suitable fish habitat conditions upstream and similarities in habitats upstream and downstream of the channels, species richness was lower upstream of both channels (Table 14-11).

In addition to declines in species richness, blockages can also alter fish abundance and density (Kenny et al. 1992). Recovery of pre-disturbance abundance and density in upstream species may be delayed as a result of a complete or intermittent downstream blockage (Detenbeck et al. 1992). Abundances of upstream species may also shift as a result of the exclusion of top predators by downstream barriers. For example, the exclusion of American eel (*A. rostrata*), a species known to prey upon benthic species such as cottids and percids (Jenkins and Burkhead 1994; Wiley et al. 2004), from much of its historic range as a result of stream blockages may partially explain high abundances of Blue Ridge sculpin (*Cottus caeruleomentum*) above many large dams in Maryland. Large dams (> 25 ft) are present in Patapsco River, Gunpowder River, and Bush River basins in the Piedmont of Maryland. These dams serve as complete or partial barriers to upstream migration of the catadromous American eel, resulting in high eel abundances and densities downstream (Wiley et al. 2004).

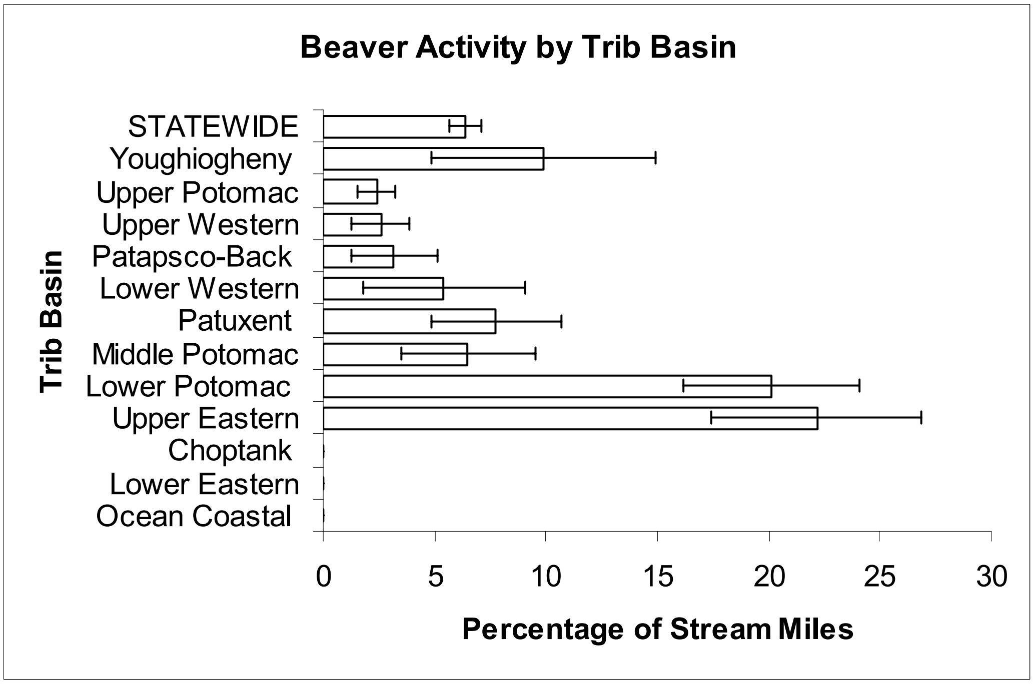


Figure 14-39. Percentage of stream miles (±SE) with beaver ponds, statewide and for the basins sampled in the 2000-2004 MBSS.

Table 14-11. Species present above and below fish blockages formed by concrete channels at sites sampled by MBSS in Bald Hill Branch and Unnamed Tributary (UT) to Southwest Branch in Western Branch basin.

Stream	Fish Species	Downstream of Barrier	Upstream of Barrier
Bald Hill Branch	Mummichog (*Fundulus heteroclitus*)	X	X
	Blacknose dace (*Rhinichthys atratulus*)	X	X
	Golden shiner (*Notemigonus crysoleucas*)	X	X
	Goldfish (*Carassius auratus*)	X	X
	Swallowtail shiner (*Notropis procne*)	X	
	Bluespotted sunfish (*Enneacanthus gloriosus*)	X	
	Redbreast sunfish (*Lepomis auritus*)	X	
	Creek chubsucker (*Erimyzon oblongus*)	X	
	Eastern mudminnow (*Umbra pygmaea*)	X	
	Pumpkinseed (*Lepomis gibbosus*)	X	
UT Southwest Branch	Blacknose dace (*Rhinichthys atratulus*)	X	X
	Least brook lamprey (*Lampetra aepyptera*)	X	X
	American eel (*Anguilla rostrata*)	X	X
	Rosyside dace (*Clinostomus funduloides*)	X	
	Fallfish (*Semotilus corporalis*)	X	
	Swallowtail shiner (*Notropis procne*)	X	
	Tessellated darter (*Etheostoma olmstedi*)	X	
	Cutlips minnow (*Exoglossum maxillingua*)	X	
	White sucker (*Catostomus commersoni*)	X	
	Common shiner (*Luxilus cornutus*)	X	
	American brook lamprey (*Lampetra appendix*)	X	
	Pumpkinseed (*Lepomis gibbosus*)	X	
	Bluegill (*Lepomis macrochirus*)	X	
	Satinfin shiner (*Cyprinella analostana*)	X	
	Redbreast sunfish (*Lepomis auritus*)	X	
	Eastern mosquitofish (*Gambrusia holbrooki*)	X	
	Creek chub (*Semotilus atromaculatus*)	X	
	Sea lamprey (*Petromyzon marinus*)	X	

An analysis of 47 MBSS sites in these basins was used to examine the possible influence of eel predation on sculpin abundance above and below large dams. Sites were selected based on the presence of preferred sculpin habitat and suitable eel habitat (Stranko et al. 2005b). Sculpin abundance was significantly higher in sites above dams compared to below (Table 14-12). In downstream sites where the two species co-existed, sculpin abundance was inversely related to eel abundance. This suggests that predation by eels may lower sculpin abundance where they co-occur. Exclusion of eels from streams above dams, therefore, results in higher sculpin abundance due to the absence of this important predator. Other variables (e.g., microhabitat) not measured by MBSS may be additional factors influencing sculpin abundance. However, the exclusion of eel from upstream habitats likely has cascading effects on fish community structure upstream of blockages.

Deleterious effects of stream blockages are not limited to stream fish communities. Barriers have also been implicated in the decline of freshwater unionid mussels throughout North America (Watters 1996). The parasitic larval stage of most unionids requires fishes as hosts. Stream barriers can indirectly result in declines of unionid populations by directly excluding host species upstream, and by altering upstream habitats such that unfavorable conditions reduce survival of host fishes (Bogan 1993). Therefore, stream blockages that serve as barriers to host fishes may cause isolation and fragmentation of unionid populations, leading to local extinctions (Watters 1992, 1996). Stream barriers may be partially responsible for the decline of rare unionids in Maryland. For example, the distribution of the federally endangered dwarf wedge-mussel (*Alasmidonta heterodon*) in certain streams is confined to stream reaches below road crossings. The downstream distribution of this species in relation to road crossings suggests that complete or partial blockages at these crossings may impede the upstream movement of anadromous and resident host fishes, thereby restricting dwarf wedgemussel to downstream habitats. The role of stream barriers in unionid distribution in Maryland requires further investigation.

A strong case can be made regarding the deleterious effects of blockages on fish communities. However, in some cases a blockage may be beneficial to upstream fish communities because it prevents the upstream introduce-tion of an invasive, non-indigenous fish species. For example, Timber Run, a tributary to Liberty Reservoir in Baltimore County, has been monitored annually as part of the MBSS sentinel site network. This tributary has supported a healthy population of native brook trout (*Salvelinus fontinalis*) due in part to a downstream blockage (road culvert) that excluded non-native brown trout (*Salmo trutta*) from upstream reaches. Since 2000, siltation below the road culvert has reduced the height of the blockage, ultimately allowing passage of brown trout into upstream habitats. Competition between brook trout and this non-native trout species may ultimately lead to the extirpation of brook trout from this tributary. In the large majority of circumstances, the removal of stream blockages will benefit stream communities. However, when planning fish passage projects in areas where non-native species are known to occur, the potential threats of non-native introductions to upstream areas should be considered.

Stressors to streams are often difficult to definitively diagnose. Disruptions to the connectivity of stream eco-systems may be responsible for stream degradation in more cases than are presently known even though extensive efforts have been made to restore access of anadromous fish species to their natal spawning areas, substantial amounts of stream habitat are still inaccessible. In particular, the distribution and effects of small blockages associated with road crossings on resident species in Maryland is not well understood and are not limited to fish and mussels.

14.4.5.2 Channelization

Channelization can dramatically change the character of a stream. Historically, streams were routinely channelized to drain fields and provide flood control. Today, streams in urban areas are often channelized to accommodate road building or to drain stormwater from developed areas. When previously meandering streams are straightened, they lose their natural connection to the floodplain, with adverse consequences for the stream ecosystem. For example, increased flows during storm events can lead to greater scouring, more bank instability, and disruption of the natural pattern of riffle and pool habitats. At other times, decreased base flows can result in stagnant ditches with substrates degraded by heavy sediment deposition. At the extreme, channelization replaces a diverse, meandering stream with a barren concrete trapezoid devoid of physical habitat.

Table 14-12. Above and below dam abundance of Blue Ridge sculpin and American eel.

Species	Above Dam (Mean abundance)	Below Dam (Mean abundance)
Blue Ridge sculpin (*C. caeruleomentum*)	260.7*	42.3*
American eel (*A. rostrata*)	0.25*	26.4*
*indicates significant difference (p<0.05)		

Figure 14-40 shows that channelized streams have lower BIBIs. This effect is widespread in Maryland (Figure 14-41). Statewide, an estimated 22% of stream miles are channelized. The greatest extent of channelization was observed in the Ocean Coastal basin (67%). Two additional basins had channelization in greater than 50% of the stream miles, Lower Eastern Shore (62%) and the Choptank (53%). The Lower Potomac River, Lower Western Shore, and the Youghiogheny had less than 10% of stream miles with channelization. All other basins had between 12-33% of stream mile channelized.

14.4.5.3 Altered Flow Regimes

As described above, increased runoff from impervious surfaces and increased flows in straightened and constrained channels can lead to greater scouring, greater bank instability, and disruption of the natural pattern of riffle and pool habitats. At other times, decreased base flows can result in stagnant ditches with substrates degraded by heavy sediment deposition. Therefore, both higher and lower than natural flows can have deleterious effects on stream biota.

Low Flows. For the years 2000-2004, 4% of streams were unsampleable due to low flows. Some of these low flows are attributable to natural conditions, but others may result from less infiltration in urban areas or from specific water withdrawals. Currently, the MDE maintains minimum flow-by requirements for surface water withdrawal permits to protect the State's aquatic resources. It is not known whether the permits are uniformly applied or whether they consider the potential cumulative effects of other permitted withdrawals on the same waterway. Preliminary analysis by the MBSS did not reveal evidence of water withdrawal effects on biological condition (see Volume 4 for more details). Figure 14-42 describes the regression relationship for catchment area and flow statewide. Certainly, different natural conditions (e.g., lower flows in rain shadow valleys and karst geology) account for most of this variability. The flow values are also one-time measurements, although most are taken during baseflow in the summer. Nonetheless, MBSS sites in the lower right corner (i.e., those departing most from the regression) may represent low flow resulting from human activities. For the six biggest outliers, FIBIs and BIBIs were not significantly lower than average.

In addition, MDE permit information in the Big Elk Creek basin was used to seek associations between the relationship of seven permitted withdrawals and eight MBSS sites (Figure 14-43). On four occasions between 1994 and 2001, stream flow in Big Elk Creek at Elk Mills dipped below 15.3 cubic feet per second (cfs), the minimum flow-by for the mill race diversion that feeds DNR's fish rearing ponds in Elkton. On two occasions, stream flow dipped below 11.9 cfs; the (waived) minimum flow-by requirement for the town of Elkton. These two low flow periods lasted for about 20 days in late summer 1995, and about 10 days in late summer 1999. An examination of the fish and benthic IBIs in the Big Elk Creek basin between 1996 and 2000, however, did not suggest that permitted surface water withdrawals operating during this time were a major stressor at these sites.

In the future, the MBSS plans to conduct a more thorough analysis with fortuitously "paired" water withdrawals and MBSS sampling sites. Stressor identification will be improved by choosing homogeneous geographic regions and focusing on especially low flow periods (e.g., drought summers). This will be possible when more of the 921 permitted surface water withdrawals are available as geographically referenced digital files.

High Flows and Bank Erosion. Bank erosion is a common symptom of stream flow problems. Erosion within the stream channel, often associated with "flashy" flow regimes in highly urbanized basins, can scour banks and mobilize sediment. In fact, much of the sediment transported and deposited within the stream often originates from in-channel erosion rather than overland flow. Bank erosion is a sign of channel instability (side-cutting and/or down-cutting). While the lack of streambank vegetation can contribute to bank erosion, severe erosion can, in turn, destabilize vegetation, causing even large tress to fall. In addition, sediments eroded from banks can become resuspended after initial settling, increasing turbidity and deposition in downstream areas. The effects of sedimentation are discussed in the next section.

Moderate to severe bank erosion occurs commonly in Maryland streams, as seen in Figure 14-44. Many basins had a high occurrence of bank erosion during the 2000-2004 MBSS. The greatest extent of moderate to severe bank erosion was estimated for the Patuxent basin (57% of stream miles). Within each 75-meter segment sampled, field estimates of the amount of eroded bank area were also made. Mean values by basin were used to estimate the extent of eroded area (square meters) per stream mile. The highest values were the Choptank (35%), Upper Eastern Shore (36%), Lower Western Shore (37%), and the Patuxent (39%) basins. Overall, moderate to severely eroded banks occurred on 23% of the State's stream miles.

As described above, channel alteration (generally) and changes to the flow regime (specifically) disrupt the natural pattern of riffles and pools in a stream. The MBSS assesses this effect with the velocity/depth diversity score. Figure 14-45 shows that lower velocity/depth diversity scores are correlated with lower FIBIs. Figure 14-46 shows the extent of this effect by tributary basin.

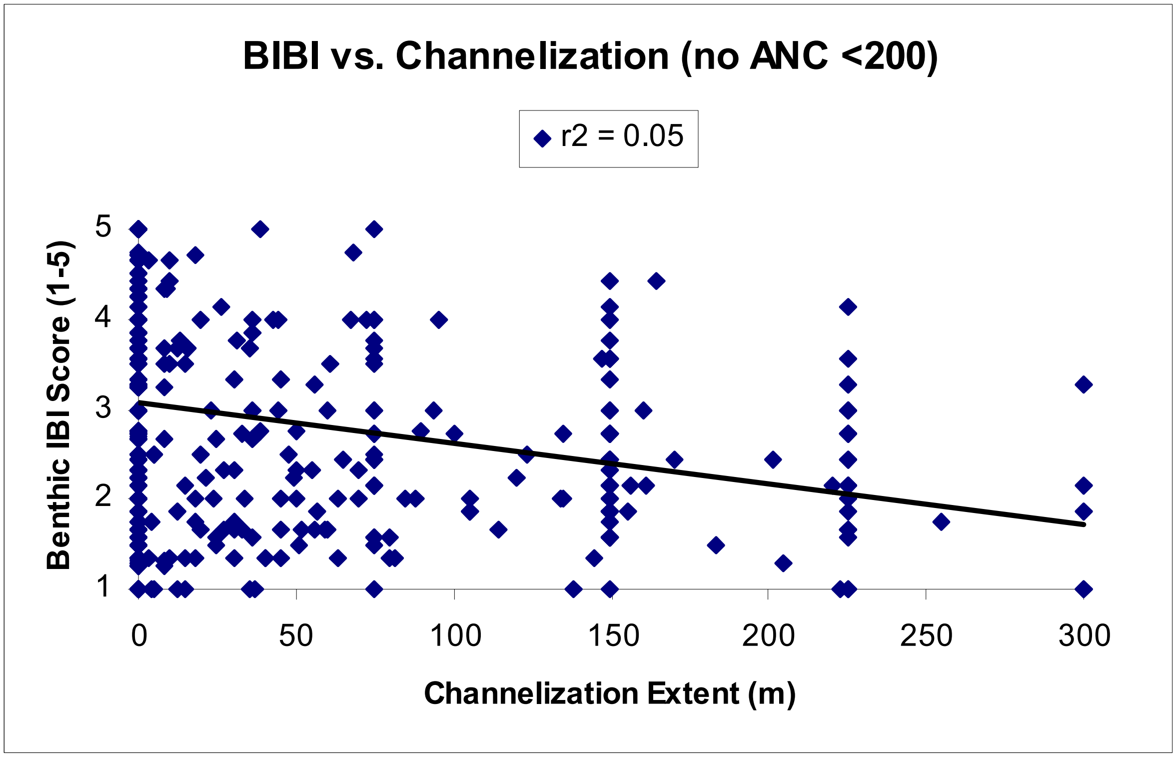


Figure 14-40. Relationship between benthic IBI and extent of channelization, statewide for the 1997-2004 MBSS sites with ANC > 200.

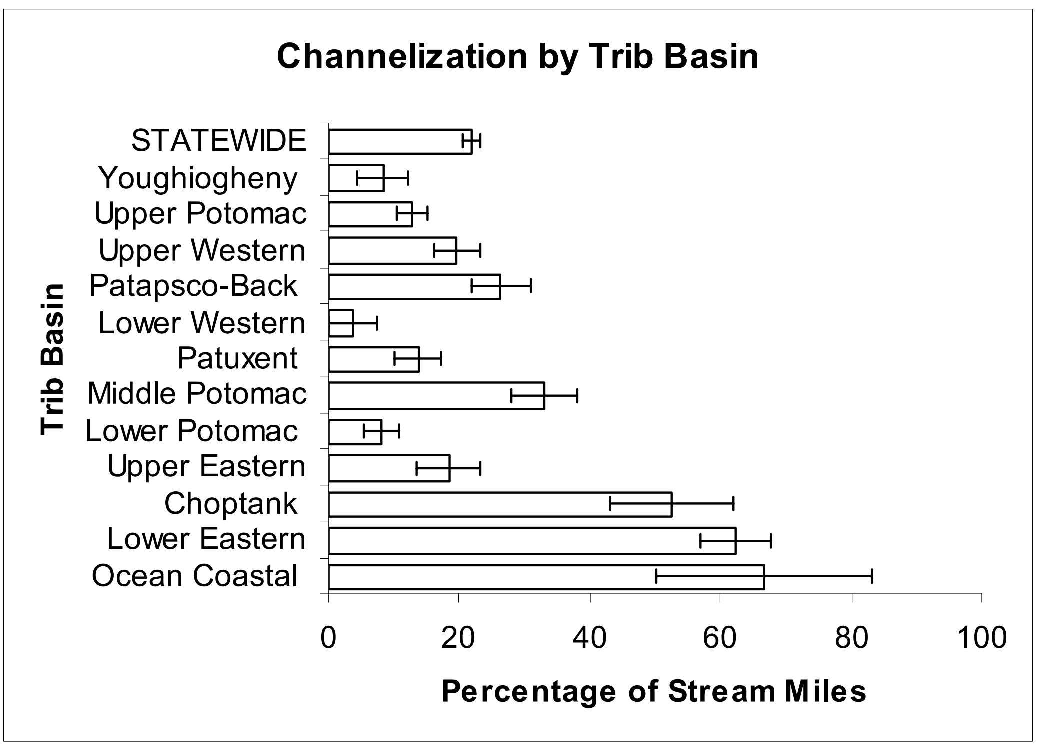


Figure 14-41. Percentage of stream miles (+SE) with evidence of channelization statewide and by basins for the 2000-2004 MBSS.

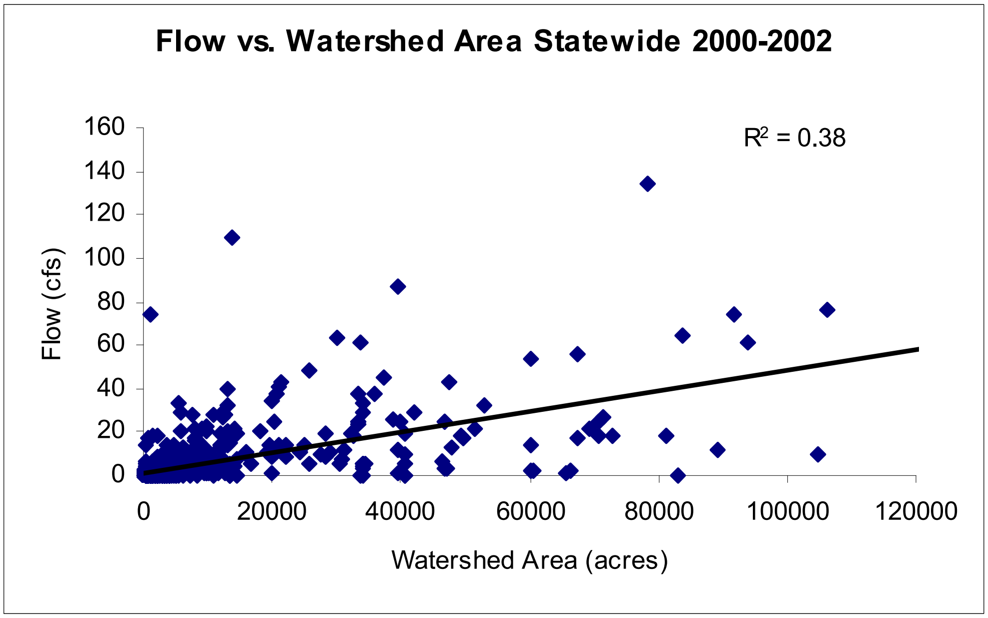


Figure 14-42. Relationship of flow and catchment (basin) area at MBSS sites (2000-2002) showing outliers with "apparent low flows."

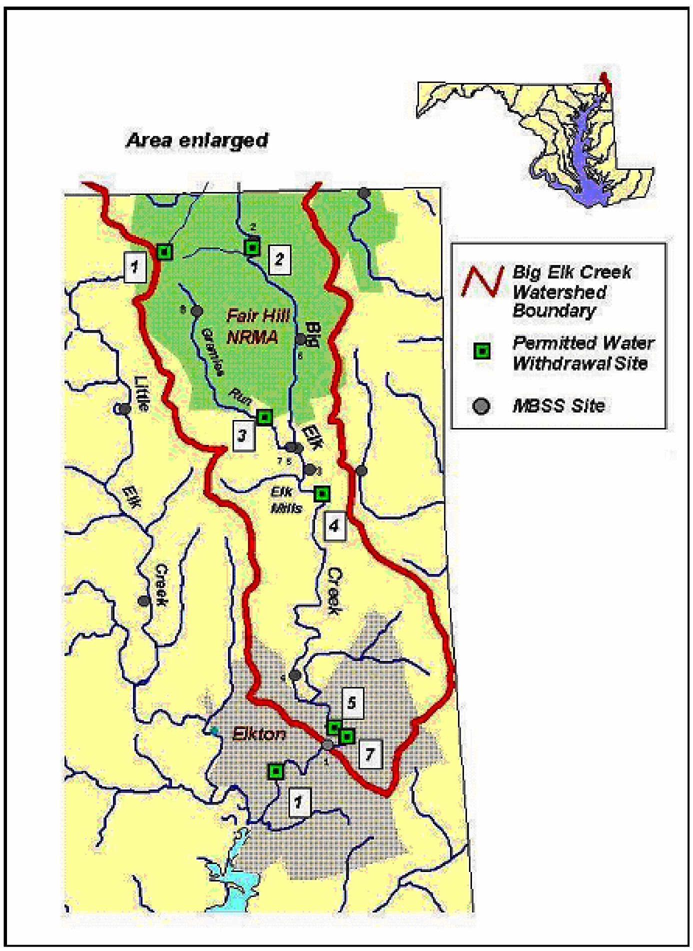


Figure 14-43. Big Elk Creek basin in Maryland showing permitted water withdrawal and MBSS sites.

14.4.6 Terrestrial and Channel Sediment

Sediment "pollution" is the number one impairment of streams nationwide. As described above, sediment within the water column and sediment within the streambed can come from both terrestrial and channel sources. Wolman (1967) described a cycle of sedimentation and erosion associated with urban development. Initially, cleared land produces large sediment loads into streams that can lead to an aggradation phase where the channels are filled with sediment. Following construction, sediment loads from the catchment are reduced and the increased high flows gradually remove the sediment so that the channel widens and deepens. During this erosional phase, most of the sediment carried by the stream comes from channel erosion rather than terrestrial sources.

Construction activities can affect aquatic biota directly and indirectly (Angermeier et al. 2004). Operating machinery in shallow-water habitats can destroy nests of animals and crush mollusks or other sedentary animals. However, more serious and common impacts result from the indirect effects of excessive fine sediment. Fine sediment can interfere with breathing, feeding, reproducing, and food production for many aquatic animals (Wood and Armitage 1997). Consequently, sediments can depress populations of invertebrates (e.g., Cline et al. 1982) and fishes (e.g., Whitney and Bailey 1959), and increase the dominance of silt-tolerant species. The deposition of fine sediments increases the embeddedness of the stream bed. Figure 14-47 shows the relationship of embeddedness with FIBI. Significant deposition of gravel and fine sediments can also lead to bar formation. Although some formation of bars is natural, severe bar formation can signal channel instability related to bank erosion and altered flow regimes. Exacerbated bar formation was observed in 48% (includes moderate bar formation in 32% of stream miles and severe in 16%) of stream miles statewide (Figure 14-48). Estimates of the percentage of stream miles experiencing moderate to severe bar formation were highest in Patuxent (62%), Lower Eastern Shore (59%), and the Lower Potomac (59%). Ocean Coastal had the lowest percent of stream miles with moderate to severe bar formation (14%). The other basins ranged between 32% and 52% for stream miles with exacerbated bar formation.

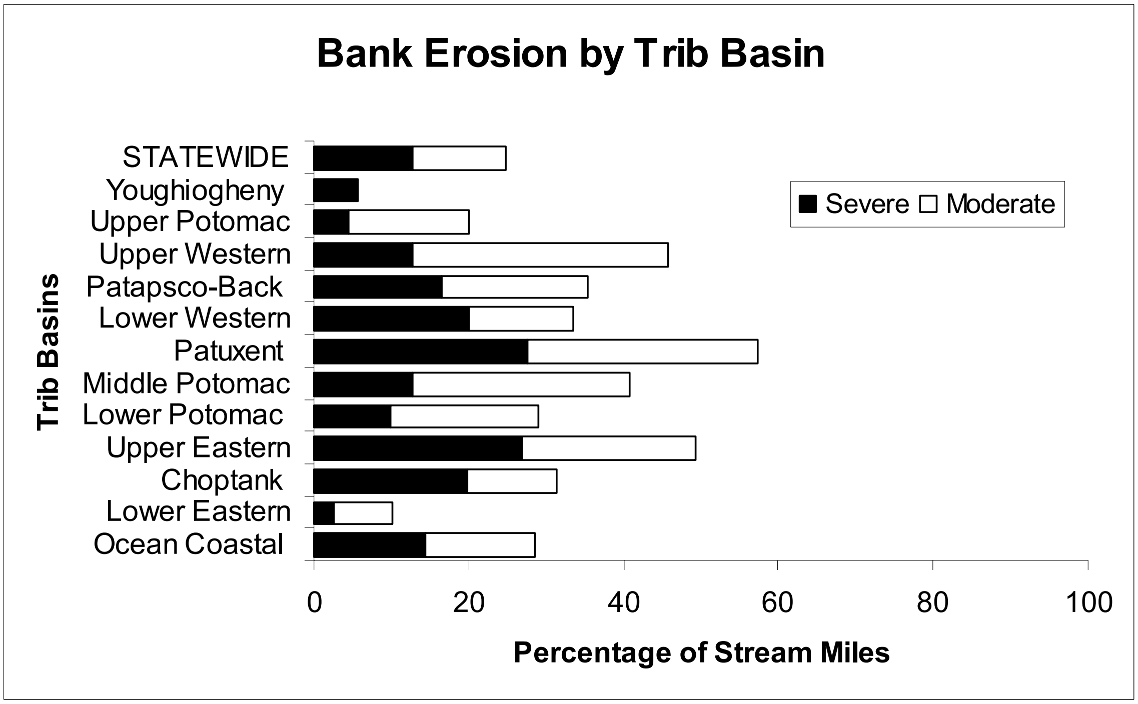


Figure 14-44. Percentage of stream miles with moderate to severe bank erosion, statewide and for the basins sampled in 2000-2004 MBSS.

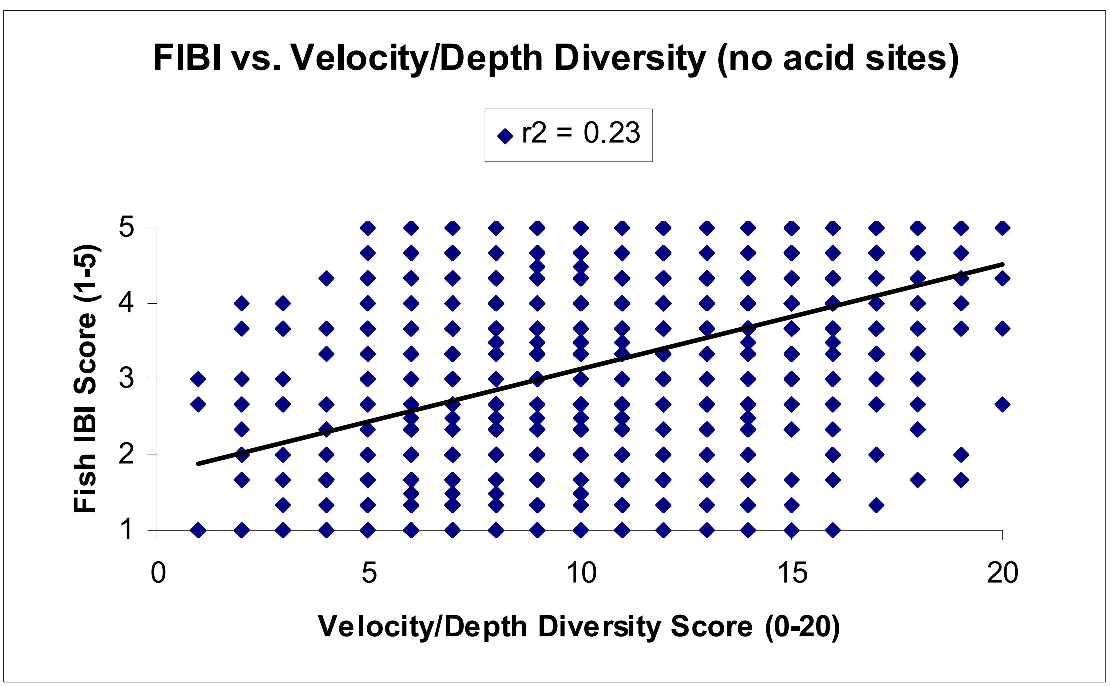


Figure 14-45. Relationship between the fish IBI and velocity/depth diversity scores, statewide for the 1995-2004 MBSS sites with ANC > 200.

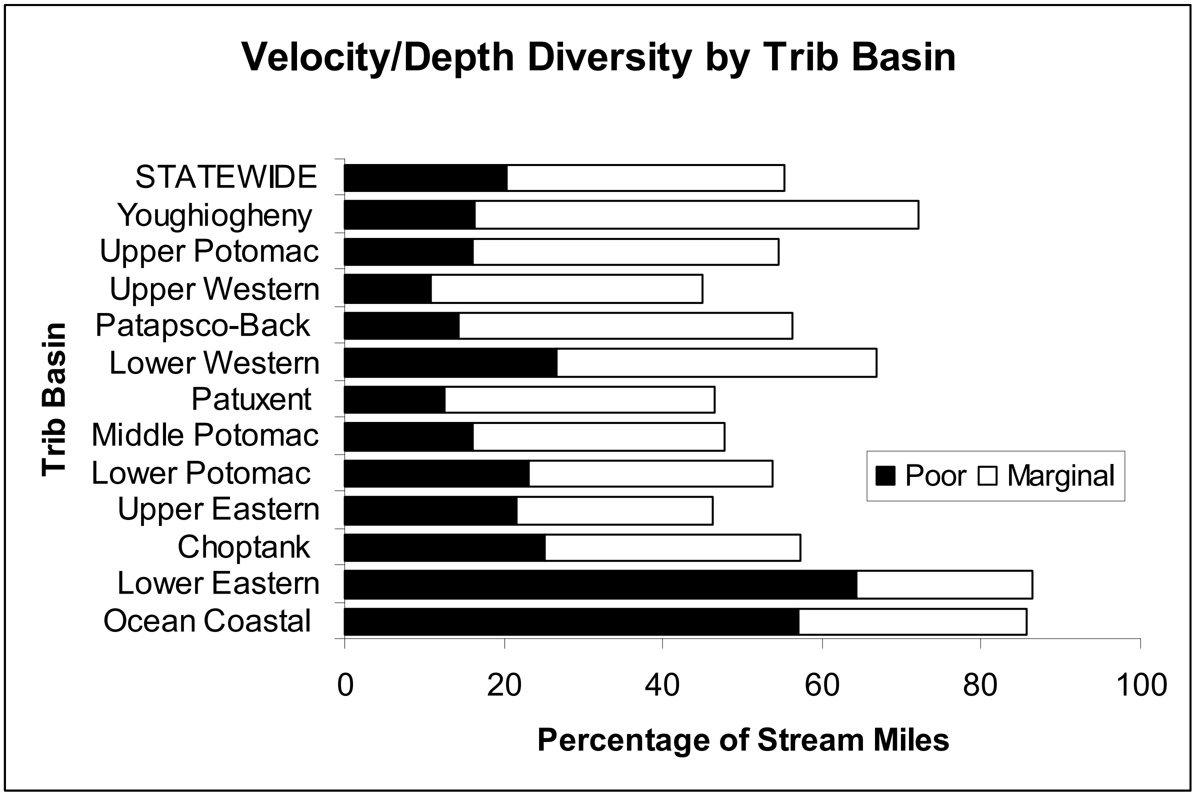


Figure 14-46. Percentage of stream miles with marginal and poor velocity/depth diversity scores, statewide and by basins sampled in 2000-2004 MBSS.

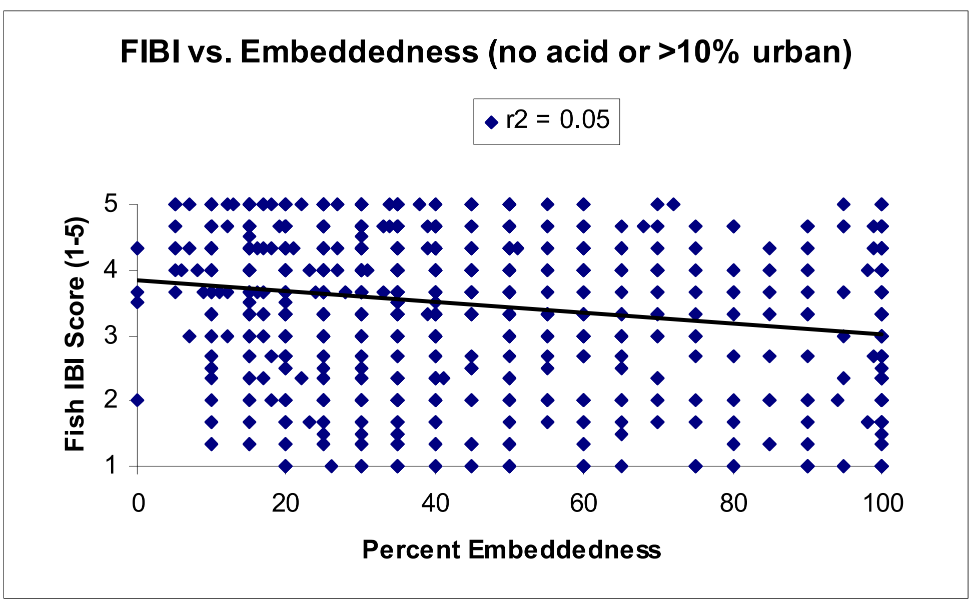


Figure 14-47. Relationship between fish IBI and embeddedness, statewide for the 1995-2004 MBSS sites with ANC > 200 and urban land < 10%.

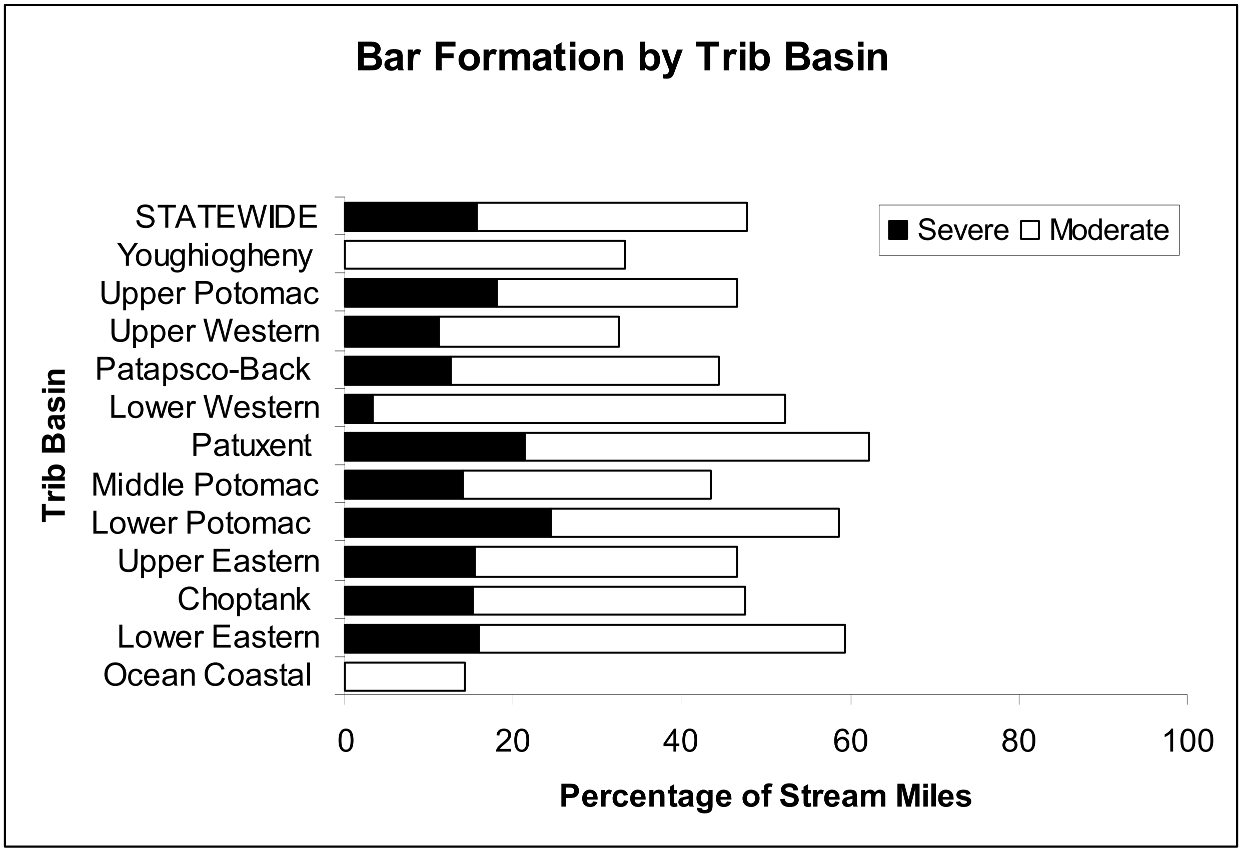


Figure 14-48. Percentage of stream miles with moderate to severe bar formation, statewide and for the basins sampled in the 2000-2004 MBSS.

SEDIMENT IMPAIRMENTS IN MARYLAND

The Maryland Department of the Environment (MDE) collaborated with Maryland DNR to develop a method for identifying sediment impairments in Maryland basins based on MBSS data. Specifically, MBSS data from 1995 to 2004 were used to identify likely sediment impairments based on sediment-related endpoints (i.e., stream habitat endpoints that best predict biological condition). This endpoint approach is consistent with U.S. Environmental Protection Agency (EPA) guidelines and uses Maryland biocriteria-based water quality standards.

To develop a model of sediment effects, the project first identified MBSS physical habitat parameters potentially influenced by sediment transport. The 27 parameters were grouped into five categories: riparian and upland zone, combined physical habitat, channel features, streambed, and water column. These parameters were reviewed and approved by a Technical Advisory Committee. Next, a subset of the parameters were selected that met the following criteria:

- Collected during both rounds of MBSS sampling (thus more sites can be analyzed)
- Useful range of values that provide discriminatory power (e.g., parameters scored as presence and absence would not meet this criterion)
- Not confounded by stream size and other critical natural variables, and
- Not completely redundant

Based on these criteria, six candidate parameters were available for analysis as potential surrogates for sediment impacts (see Table A).

The dataset was then refined by removing all MBSS sites affected by known, non-sediment related stressors (urban land use, high chloride levels, and acidification). Lastly, logistic regression was used to develop the best model (sediment indicator) for identifying sediment-related effects on "biocriteria failure" (i.e., degraded biological communities as represented by fish and benthic IBI scores). The modeling procedure is illustrated in Figure A.

Logistic regression models were developed at both the state and regional scale (i.e., for the Highlands, Eastern Piedmont, and Coastal Plain). Half of the original dataset was used for model development (based on random selection of stations) and the remaining half was used for model validation. Three to four parameters were selected per region based on model scores and parsimony (Table B). Table C provides the average rate of correct classification (ARCC) for both the development and validation results. MDE is evaluating different options for applying this sediment indicator at the Maryland 8-digit basin or other spatial scales. This model and methodology are currently under review and thus all results are draft.

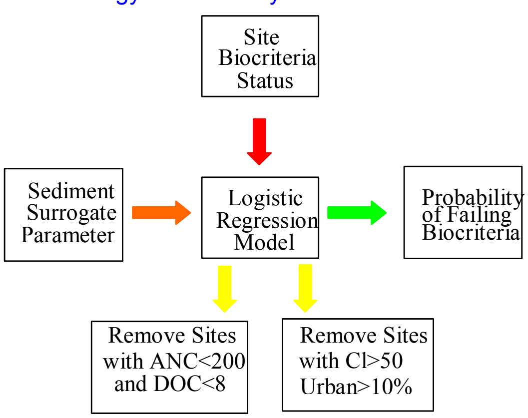


Figure A. Flow chart of logistic regression approach to developing a statistical model (sediment indicator) for predicting biocriteria failure.

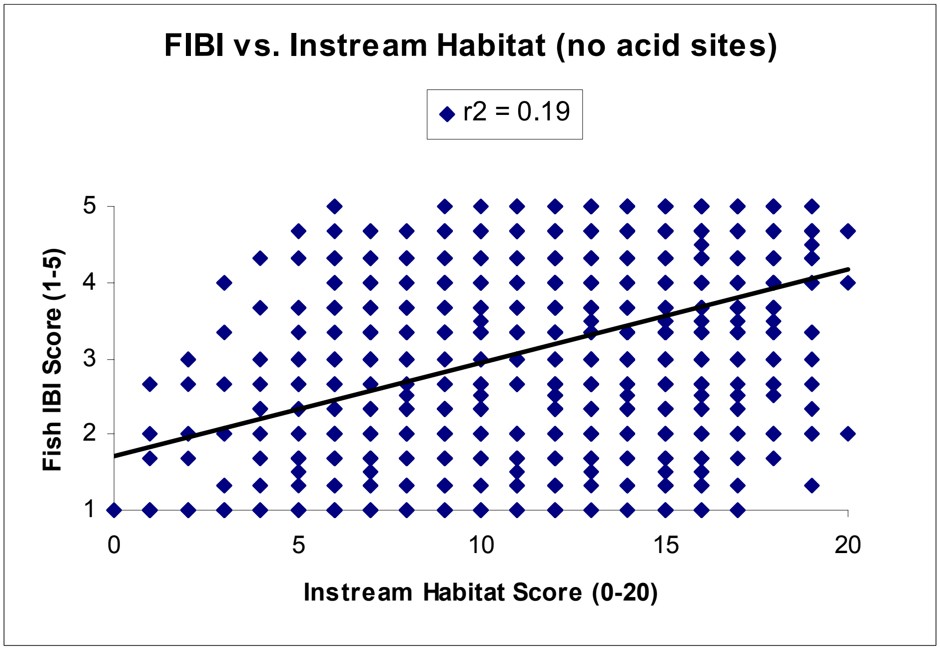
Table A. Six candidate parameters with identified relationships to sediment that may serve as useful surrogates for predicting stream impairment.			
Surrogate Variables	**Definition**	**Scoring**	**Relationship to Sediment**
Riffle/Run Quality	Visual rating based on the depth, complexity, and functional importance of riffle/run habitat, especially deeper riffle/run areas, stable substrates, and a variety of current velocities.	0 to 20	High quality riffle/run habitat is evidence of lack of sediment deposition. However, riffle/run quality is confounded by natural variability (i.e., some streams will naturally have different quality riffle/run habitat).
Bank Stability	Composite score combining visual rating based on the presence or absence of riparian vegetation and other stabilizing bank materials, such as boulders and rootwads, with quantitative measures of erosion extent and erosion severity.	0 to 100	Bank stability is evidence of lack of channel erosion, a major source of downstream sediment transport. Sediment loading may still occur through overland runoff.
Riparian Buffer Width	Width of vegetated (i.e., grass, shrubs, or trees) riparian buffer, estimated to a maximum distance of 50 meters from the stream channel.	0 to 50	Wide and well-vegetated riparian buffers are indirectly related to sedimentation as buffers remove sediment in runoff and protect banks from erosion. Riparian buffers also benefit aquatic communities by reducing stream temperature through shading, an effect unrelated to sediment.
Instream Habitat	Visual rating based on the perceived value of instream habitat to the fish community, including multiple habitat types, varied particle sizes, and uneven stream bottom.	0 to 20	High instream habitat scores are evidence of lack of sediment deposition. However, instream habitat is confounded by natural variability (i.e., some streams will naturally have more or less instream habitat).
Epifaunal Substrate	Visual rating based on the amount and variety of hard, stable substrates usable by benthic macroinvertebrates.	0 to 20	High epifaunal substrate scores are evidence of lack of sediment deposition. However, epifaunal substrate is confounded by natural variability (i.e., some streams will naturally have different kinds of epifaunal substrate).
Embeddedness	Percentage of gravel, cobble, and boulder particles in the streambed that are surrounded by fine sediment.	0 to 100	High embeddedness is direct evidence of sediment deposition. However, embeddedness is confounded by natural variability (e.g., Coastal Plain streams will naturally have more embeddedness than Highlands streams).

Table B. Significance of parameters and model predictive power (c)				
Parameter	**Highland**	**Piedmont**	**Coastal**	**Statewide**
Intercept	0.4110	0.1157	<0.0001	<0.0001
Riffle run	0.0194	-----	<0.0001	0.0003
Riparian width	0.0413	0.1306	0.0906	0.0016
Embeddedness	0.0006	0.1350	-----	0.0110
Instream habitat	-----	0.4332	0.0004	<0.0001
Epifaunal substrate	-----	0.1104	-----	-----
c (area ROC)	0.7	0.6	0.8	0.8

Table C. Model Average Rate of Correct Classification (ARCC) at 90% confidence interval					
	Rate of Correct Classification – Fail	**Rate of Correct Classification – Pass**	**Inconclusive**	**ARCC**	**Validation ARCC**
Highland	72%	78%	39%	74%	67%
Coastal	74%	88%	27%	78%	73%
State	73%	71%	23%	73%	67%

14.4.7 Habitat Quality

Sedimentation, channel alteration, low flow, and other kinds of physical habitat degradation are reflected in the instream habitat and epifaunal substrate scores at MBSS sites. Both are qualitative measures of the total available habitat preferred by fish (instream habitat) and benthic macroinvertebrates (epifaunal substrate). Figure 14-49 shows the relationship of instream habitat to FIBI and epifaunal substrate to BIBI. The extent of the effect is shown in Figures 14-50 and 14-51.



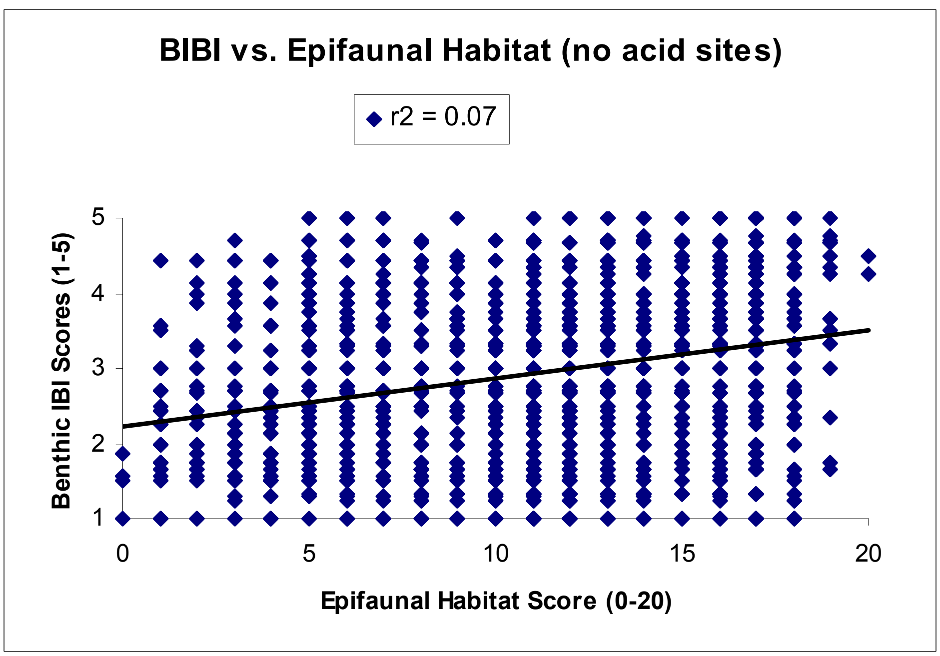


Figure 14-49. Relationship between the fish IBI and instream habitat, and benthic IBI and epifaunal substrate, statewide for the 1995-2004 MBSS.

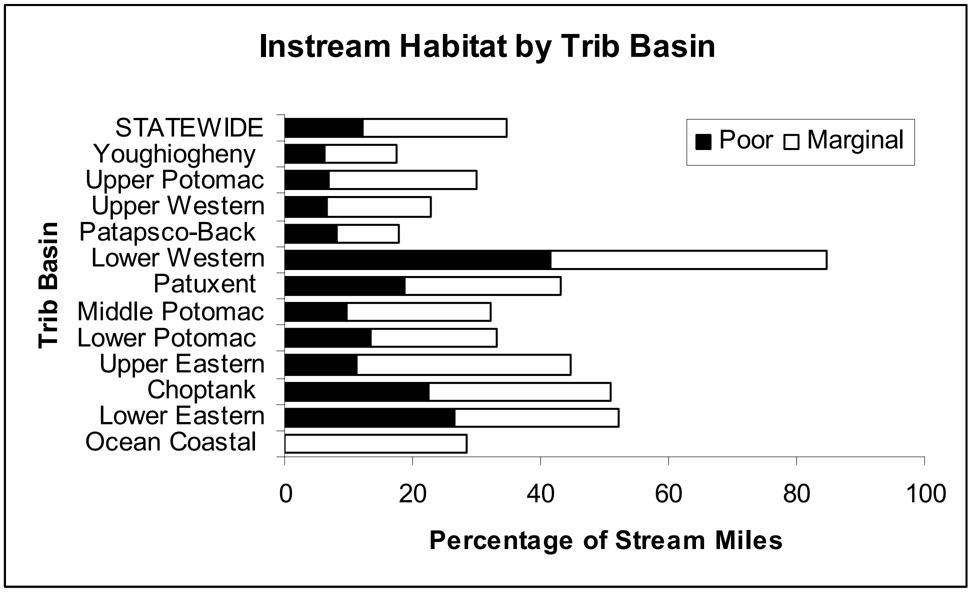


Figure 14-50. Percent of stream miles with marginal and poor instream substrate, statewide and for the basins sampled in the 2000-2004 MBSS.

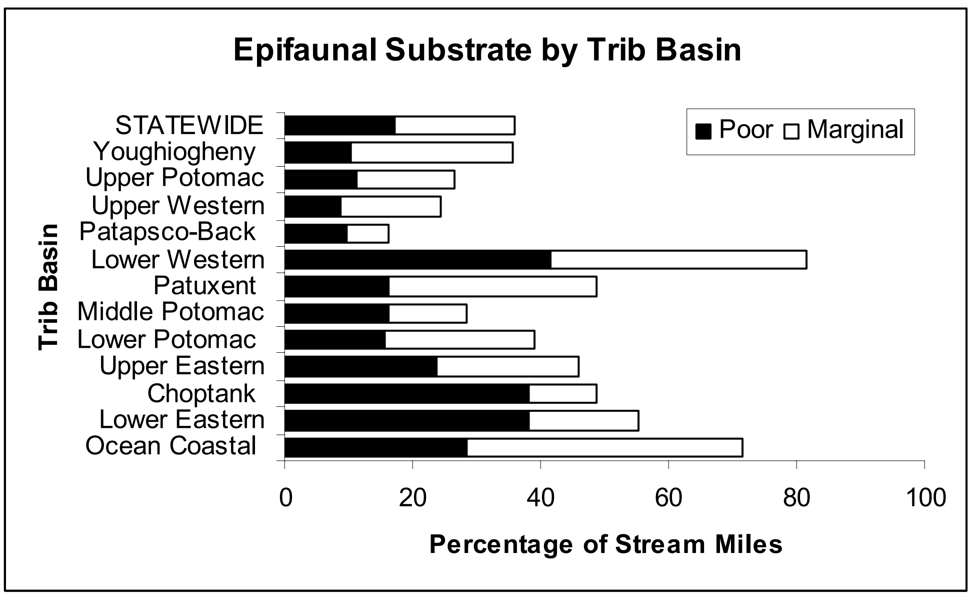


Figure 14-51. Percent of stream miles with marginal and poor epifaunal substrate, statewide and for the basins sampled in the 2000-2004 MBSS.

14.5 BIOTIC INTERACTIONS (*NON-NATIVE AND INVASIVE AQUATIC BIOTA*)

The intentional or unintentional introductions of non-native biota into streams and rivers pose potential risks to native assemblages. These risks include disease introduction, parasitism, and elimination of native species through predation, resource competition, and incidental harvest (bycatch). Table 14-13 lists the non-native species intentionally and unintentionally introduced into Maryland waters. Adverse effects can be severe, though for most species there is little documented evidence of impact to native assemblages. More details on the effects of non-native and invasive species (both aquatic and terrestrial) are provided in Volume 12: Stream and Riverine Biodiversity.

Based on historical data and survey work done by the MBSS, several non-native species of crayfish appear to be expanding in Maryland. The expansion of introduced crayfish and concomitant loss of two native crayfish from the same area strongly support the possibility of competitive or predatory exclusion. Available literature also indicates that introduced crayfish may play an important role in the elimination of freshwater mussels by preying on juveniles. The spread of non-native crayfish to additional basins in Maryland could have disastrous results for the remaining freshwater mussels in the state. The expansion of *Corbicula*, an introduced Asian clam, throughout Maryland waters may also be having a deleterious effect on native mussels.

Among the fishes, there is potential for high profile species such as northern snakehead to reduce or eliminate native fish populations. There is also a continuing risk from fish species stocked for recreational fishing, including brown trout. During 2001-2003, DNR Fisheries stocked approximately 5.4 million non-native fish, or about 1.8 million fish per year, to provide recreational fishing opportunities or enhance the forage base for gamefish (Rivers, personal communication). Eleven different species have been stocked, with life history stages ranging from adult to fry (Table 14-13). In addition, an unknown number of non-native fish and crayfish used as bait are released by anglers.

The risk of spreading non-native diseases to native biota via the culture of hatchery fish is currently unknown. A decade ago, the risk of transferring hatchery diseases to wild fish was thought to be very minimal, but the decimation of native rainbow trout in many areas of the western U.S. by whirling disease suggests that the potential for disease impacts exists. Diseases confirmed at one or more coldwater hatchery facilities or in their receiving streams during 2000-2004 included: the parasite *Ichthyophthirius multifilus* (Ich), bacterial gill disease, Columnaris, and whirling disease caused by the parasite *Myxobolus cerebralis* (Rivers, pers. comm.). It should be noted that no hatchery diseases have been determined to had a negative effect on native biota in Maryland, although no monitoring of native species for this purpose is conducted. The number of types of disease introduced from bait shops is unknown.

Results from MBSS surveys show that non-native species occur widely in Maryland. Five of the six most abundant non-native fish species found by the MBSS in 2000-2004 were members of the sunfish family. The most abundant was bluegill, followed by green sunfish, smallmouth bass, brown trout, largemouth bass, and rock bass. Another species, rainbow darter, has expanded in Maryland, and now occurs in four major basins. There is a well-established body of published literature on the negative effects of brown trout on brook trout (Fausch and White 1981; Waters 1983; Lasenby and Kerr 2001), Maryland's only native salmonid and a species designated as being in Greatest Conservation Need (GCN). Further, the lack of abundance of brook trout when brown trout were present at MBSS sites supports the concept that introduced brown trout can eliminate or greatly reduce brook trout populations (see more details in the following subsection).

14.5.1 Non-native Brown Trout as Stressors to Brook Trout in Maryland

Historically, brook trout probably inhabited much of Maryland, west of the Fall Line between the Piedmont and Coastal Plain, as well as the western border of the Coastal Plain (Boward et al. 1999). Currently, brook trout are restricted to relatively small streams (Morgan et al. 2005) and primarily occur in the westernmost portion of the State. Urban development and other abiotic variables have been shown to be associated with brook trout declines in Maryland (Morgan et al. 2005, Boward et al. 1999). However, biotic interactions with non-native salmonids have also likely influenced the current distribution of brook trout. Negative impacts of non-native salmonids on native brook trout populations have been demonstrated in other areas of eastern North America due to competition (Fausch and White 1981, Nyman 1970, Essington et al. 1998) and possibly predation (Nyman 1970).

First introduced into Maryland during the late 1800s, brown trout have become widely established in Maryland (Stinefelt et al. 1985). At present, the geographic distribution significantly overlaps that of the native brook trout (Figure 14-52). The two species prefer similar stream habitats that are dominated by cool water and abundant cover. In Maryland, it appears that many subpopulations of brook trout have become isolated into small tributaries as a result of brown trout in nearby stream reaches (Morgan et al. 2005).

Table 14-13. Non-native freshwater fish species known or thought to occur in Maryland.

Species	Status	Habitat and Extent	Impacts to Native Species
Brown trout	>50,000 stocked annually by DNR; numerous reproducing populations exist	Widely distributed in cool and coldwater habitats	Well documented impacts to brook trout; possible impacts to non-game fishes; possible disease introduction
Rainbow trout	>500,000 stocked annually by DNR; only two reproducing populations known to exist (Hoyes Run and Sang Run)	Widely distributed in cool and coldwater habitats; put and take stocking in Coastal Plain areas as well	Well documented impacts to brook trout; possible impacts to non-game fishes; possible disease introduction
Cutthroat trout	Periodically stocked by DNR; only two reproducing populations known to exist (Jennings Randolf tailrace and Murley Run)	Mostly restricted to North Branch Potomac River	Possible impacts to non-game fishes; possible disease introduction
Lake trout*	Last stocked in 1986, few reported caught in recent years	Stocked in Jennings Randolph Reservoir; not reported from outside the impoundment	None documented
Channel catfish	Reproducing populations known from most major Bay tributaries.	Widely distributed in impoundments, larger rivers, oligohaline water	Possible negative impact to white catfish
Blue catfish*	Reproducing population in the tidal Potomac River	Appears to prefer tidal oligohaline water	None documented
Flathead catfish*	Known from the Susquehanna River in Pennsylvania	Large river and impoundment species	None documented
Northern pike+	Reproducing population known from Deep Creek Lake; also stocked in other impoundments	Primarily impoundments; spawns in flooded wetlands and inlet streams	Predation likely on GCN species that occupy same habitat
Tiger muskie*	About 25,000 stocked annually by DNR, including Potomac River; hybrids are sterile	Large river and impoundment species	Predation likely on GCN species that occupy same habitat; potential disease introduction
Fathead minnow	Common bait fish species, stocked as forage by DNR; reproducing populations in some streams	Small-medium streams	None documented; may supplant native species in highly disturbed habitats; possible disease introduction
Goldfish	Sold as bait, also commonly released as pets; reproduce in ponds, reservoirs, and larger streams/rivers	Slow water habitat	None documented; possible disease introduction
Common carp	Introduced in 1870s; widespread reproducing populations	Slow water habitat in larger streams, rivers and impoundments	None documented
Grass carp*	Sold as SAV control for golf course ponds, etc. Likely in scattered ponds throughout the state	Slow water habitat in larger streams, rivers, and impoundments	None documented but pose significant threat to SAV; possible disease introduction from illegal shipments
Northern snakehead*	Released into Potomac River and Crofton Ponds from pet trade, food trade, and/or religious purposes; possible population in tidal Potomac	Slow water habitat in larger streams, rivers, and impoundments	None documented; possible disease introduction from illegal stocking
Banded darter*	Introduced into Susquehanna River; reproducing populations in MD; apparently declined in last several decades	Larger streams	None documented

Table 14-13. (Continued)			
Species	**Status**	**Habitat and Extent**	**Impacts to Native Species**
Rainbow darter	Likely introduced into MD portion of Potomac drainage; distribution expanding	Run habitat in larger streams and rivers	None documented
Walleye*	About 800,000 stocked annually by DNR, including Potomac River	Larger streams, rivers, and impoundments	None documented, but predation on native rare cyprinids likely; possible disease introduction
Largemouth bass	Introduced to MD in 1870s; now statewide reproducing populations	Slow water habitat in larger streams, rivers, ponds, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
Smallmouth bass	Introduced to Atlantic slope; widespread reproducing populations throughout non-Coastal Plain	Medium and larger streams, rivers, and impoundments with coolwater habitat	Likely impacts to smaller non-game species, including GCN species
Bluegill	Introduced to Atlantic slope; widespread reproducing populations throughout MD	Slow water habitat in larger streams, rivers, ponds, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
Rock bass	Introduced to Atlantic slope; widespread reproducing populations throughout non-Coastal Plain	Rocky habitat in streams, rivers, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
Green sunfish	Introduced to Atlantic slope; reproducing populations throughout non-Coastal Plain	Slow water habitat in streams, rivers, ponds, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
Longear sunfish	Introduced to Potomac River	Slow water habitat in larger streams and rivers	None documented
Black crappie	Introduced to Atlantic slope; widespread reproducing populations throughout MD	Slow water habitat in larger streams, rivers, ponds, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
White crappie*	Introduced to Atlantic slope; widespread reproducing populations throughout MD	Slow water habitat in larger streams, rivers, ponds, and larger impoundments	Likely impacts to smaller non-game species, including GCN species
Redear sunfish*	Introduced via pond stocking	Slow water habitat in larger streams, rivers, ponds, and larger impoundments	Potential impacts to smaller non-game species, including GCN species
* Not collected during the 2000-2004 MBSS, but known or thought to occur in MD + Collected during rare species survey in the Potomac River, 2004			

Morgan et al. (2005) also showed that streams containing brook trout and non-native species had significantly lower densities of brook trout compared to brook trout-only streams. Overall brook trout abundance in Maryland was negatively correlated (Pearsons r = -0.173, p = 0.001) with brown trout abundance (Figure 14-53) and no brook trout has ever been collected at any MBSS site with more than about 0.5 brown trout per meter of stream. The highest abundances for brook trout and brown trout occurred at sites where the other species was absent. The apparent exclusion of brook trout from sites where brown trout have become established could be due to competitive exclusion or to inherent differences in the tolerances of the two species to natural and human-related stream conditions.

Brown trout occur in streams that are significantly (ANOVA, $p < 0.0001$) more biologically (Figure 14-54), physically (Figure 14-55), and chemically (Figure 14-56) degraded than do brook trout. Brown trout also tolerate significantly warmer temperatures (Figure 14-57) and larger amounts of basin urbanization (Figure 14-58). These results suggest that degradation of streams leads to a concomitant decline in brook trout with brown trout

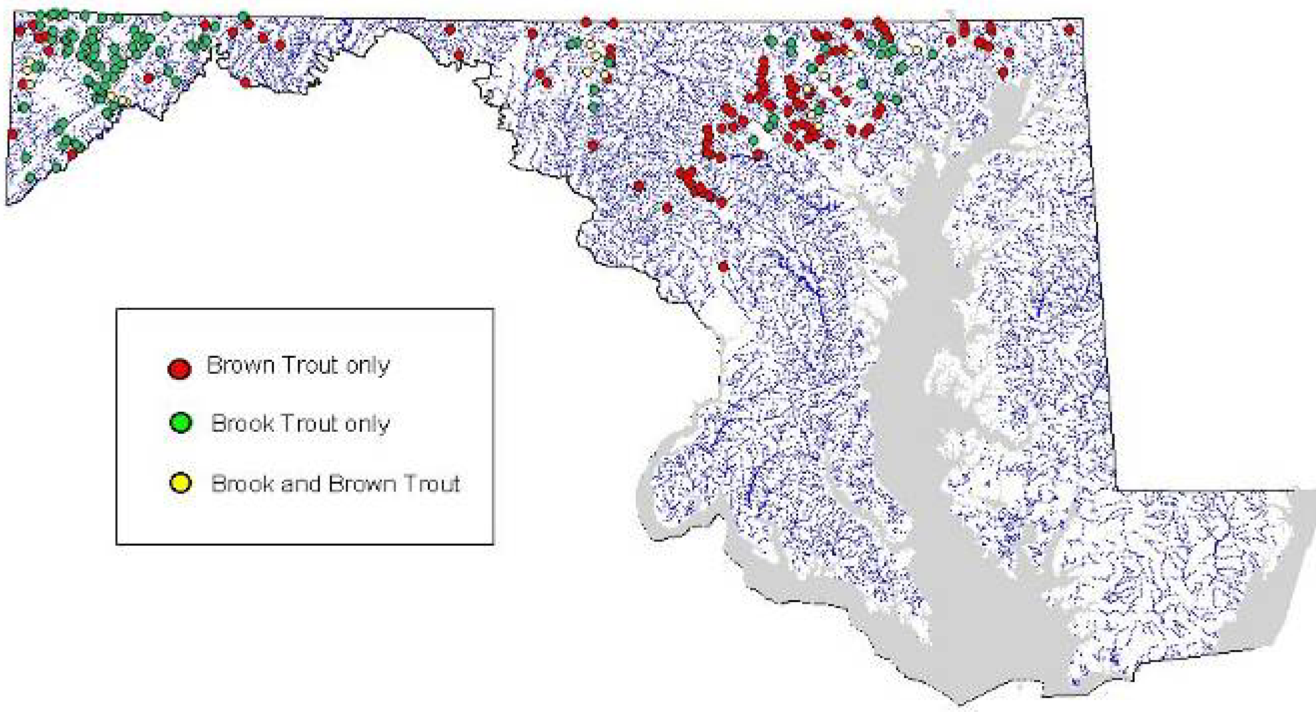


Figure 14-52. Location of sites where brook trout, brown trout, or both species were encountered in Maryland during MBSS 1994-2004.

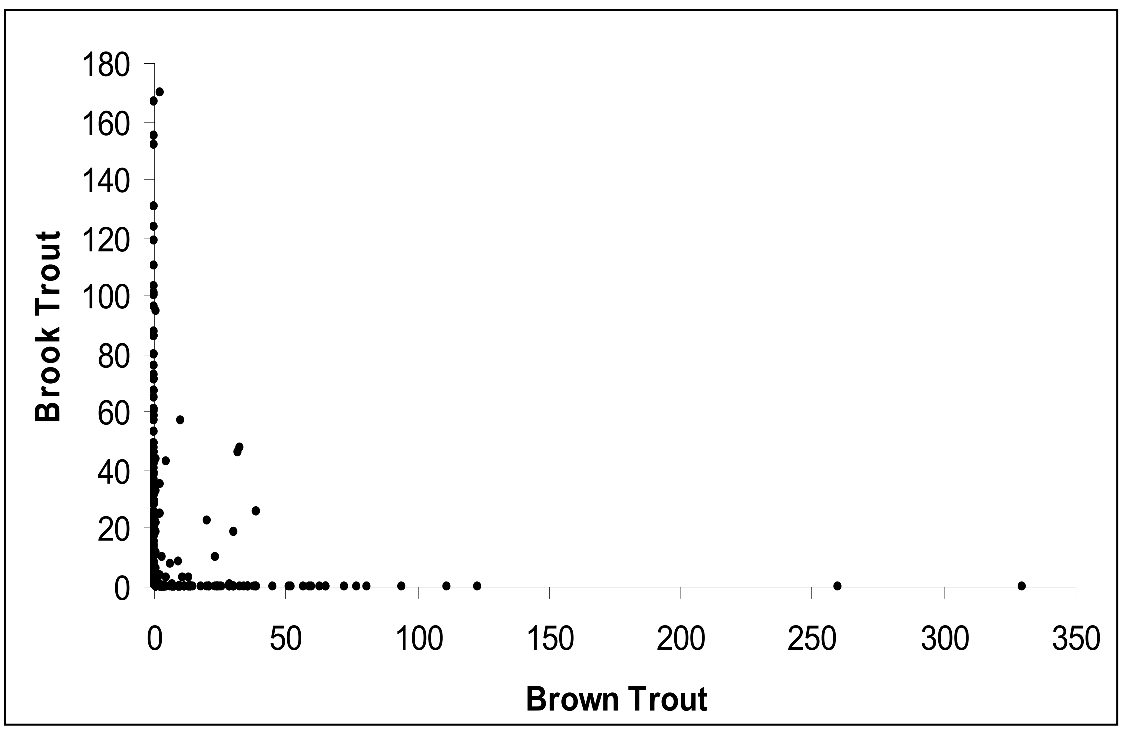


Figure 14-53. Brook trout abundance vs. brown trout abundance at 75 m stream sites for MBSS 1994-2004.

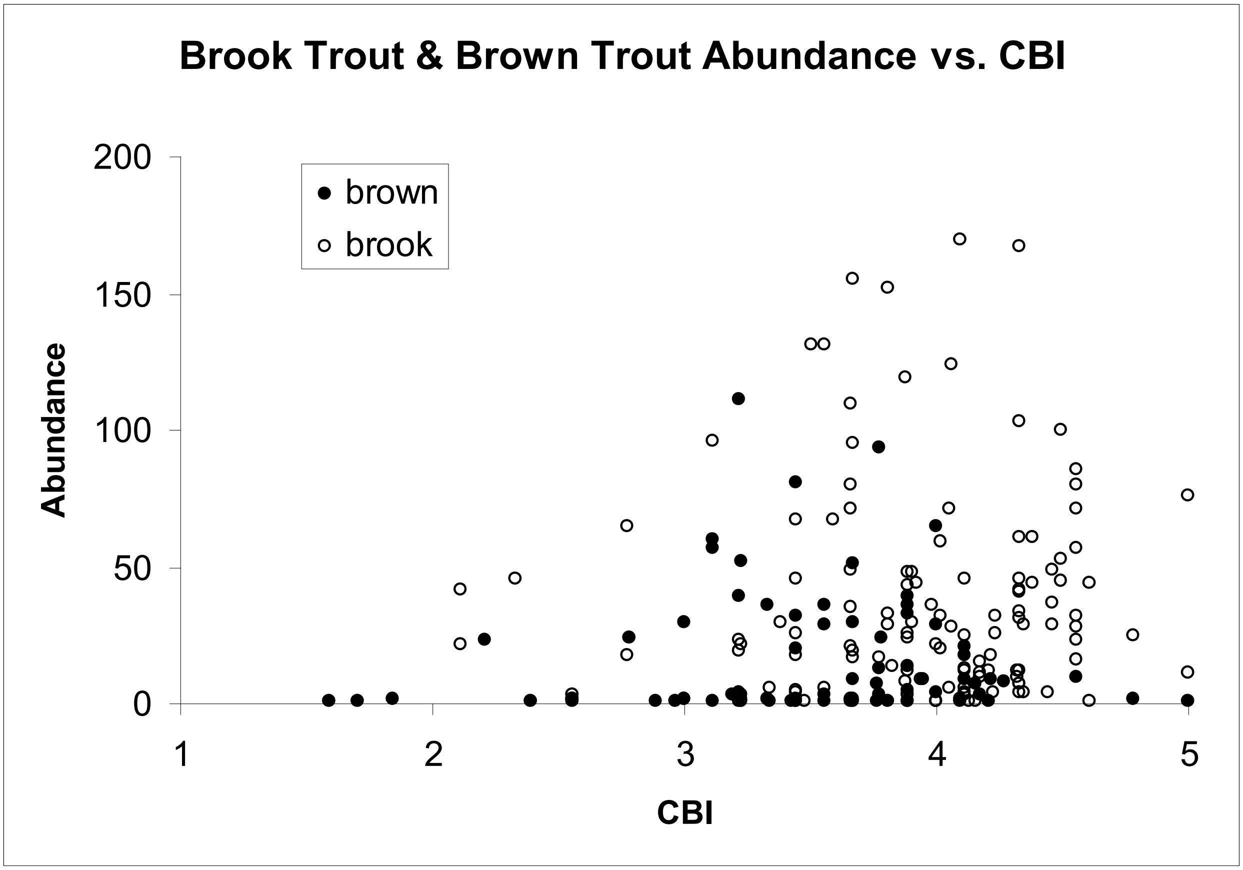


Figure 14-54. Combined Biotic Index (CBI) scores of sites containing brook trout or brown trout for MBSS 1994-2004.

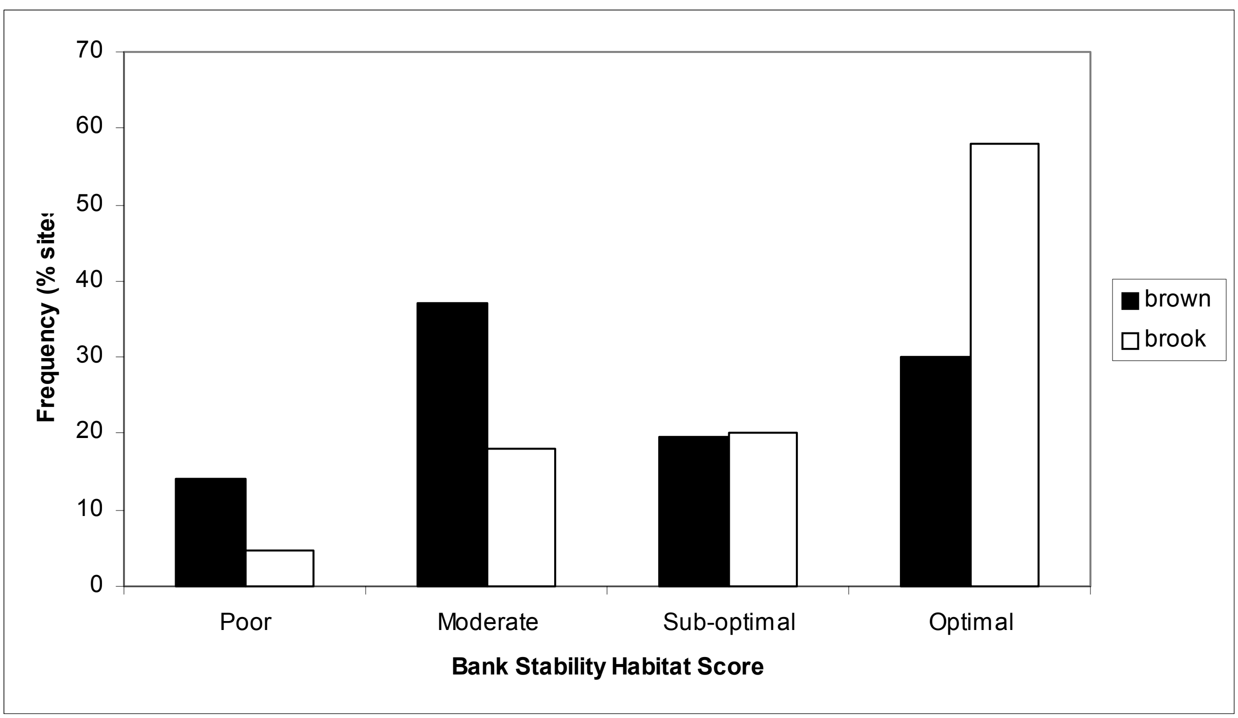


Figure 14-55. Bank stability scores of sites containing brook trout or brown trout for MBSS 1994-2004.

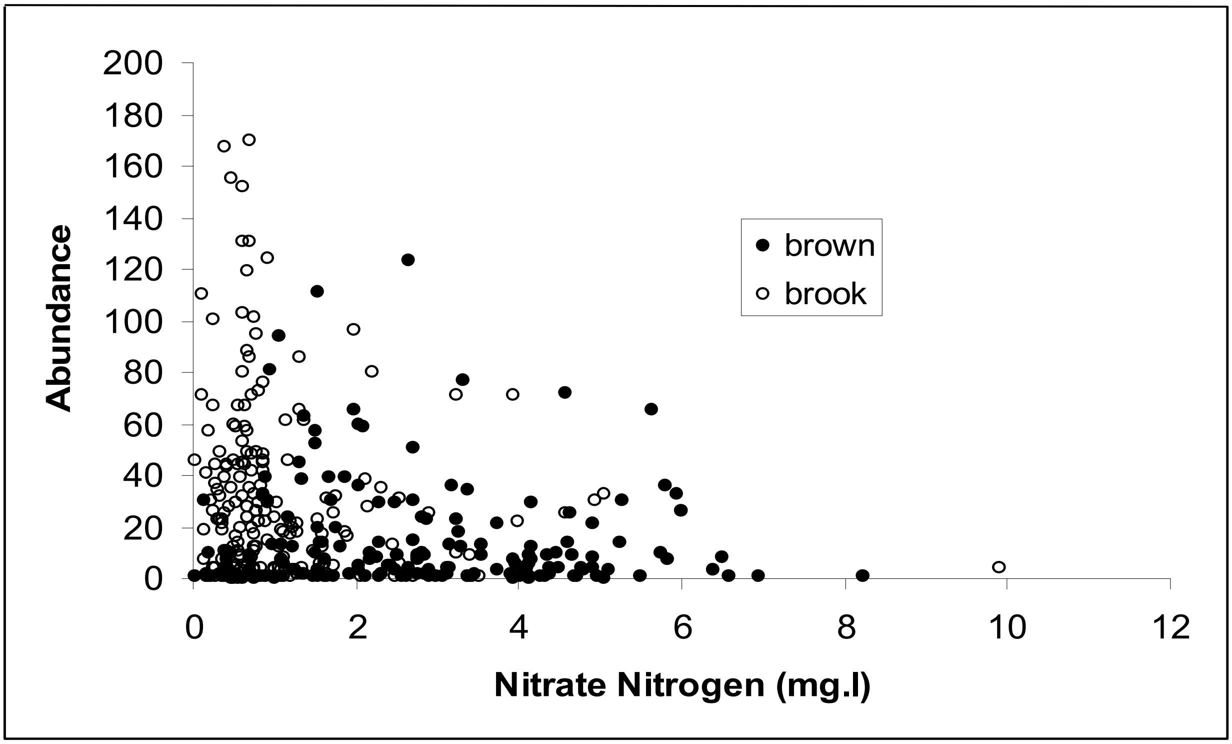


Figure 14-56. Nitrate measurements of sites containing brook trout or brown trout for MBSS 1994-2004.

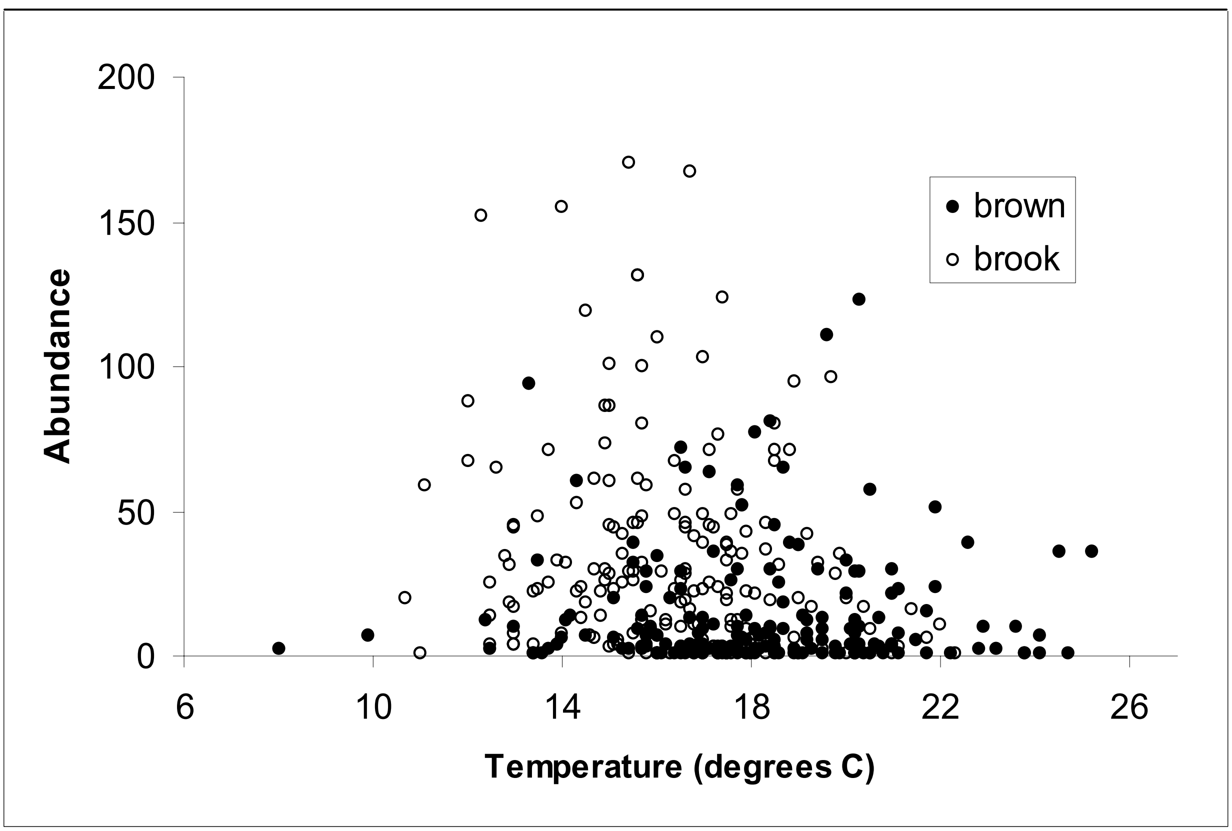


Figure 14-57. Summer water temperature (ºC) of sites containing brook trout or brown trout for MBSS 1994-2004.

gaining a competitive advantage. Table 14-14 lists thresholds for the five variables listed above at which brook trout appear to be excluded from Maryland Streams but where brown trout remain.

In addition to having a competitive advantage over brook trout in degraded streams, brown trout also appear to exclude brook trout from most streams draining more than 16,000 hectares of basin area even in the absence of major anthropogenic stressors. One exception is the Savage River in Western Maryland, which is heavily forested and remains cool throughout the summer (typically less than 18 ºC). It currently supports a fairly large brook trout population along with introduced brown trout. The only streams where brook trout presently occur in the absence of brown trout are those draining catchments of less than 14,000 hectares. Brook trout residing in these small, isolated stream reaches may still be at risk of extirpation, however, because the populations in most small reaches may be too small for long-term genetic sustainability, even if abiotic conditions in the streams remain suitable and brown trout remain excluded.

Table 14-14. Maximum thresholds where brook trout and brown trout have been collected in the MBSS data set (1994-2004).

Variable	Brook Trout Thresholds	Brown Trout Thresholds
Temperature (^{o}C)	23.8	25.2
Bank Stability Score	2	0
Urban Land Cover (%)	10.9	40.7
Nitrate-Nitrogen (mg/l)	9.9	8.2
CBI Score *	2.1	1.6
* CBI score reflects an average of FIBI and BIBI scores		

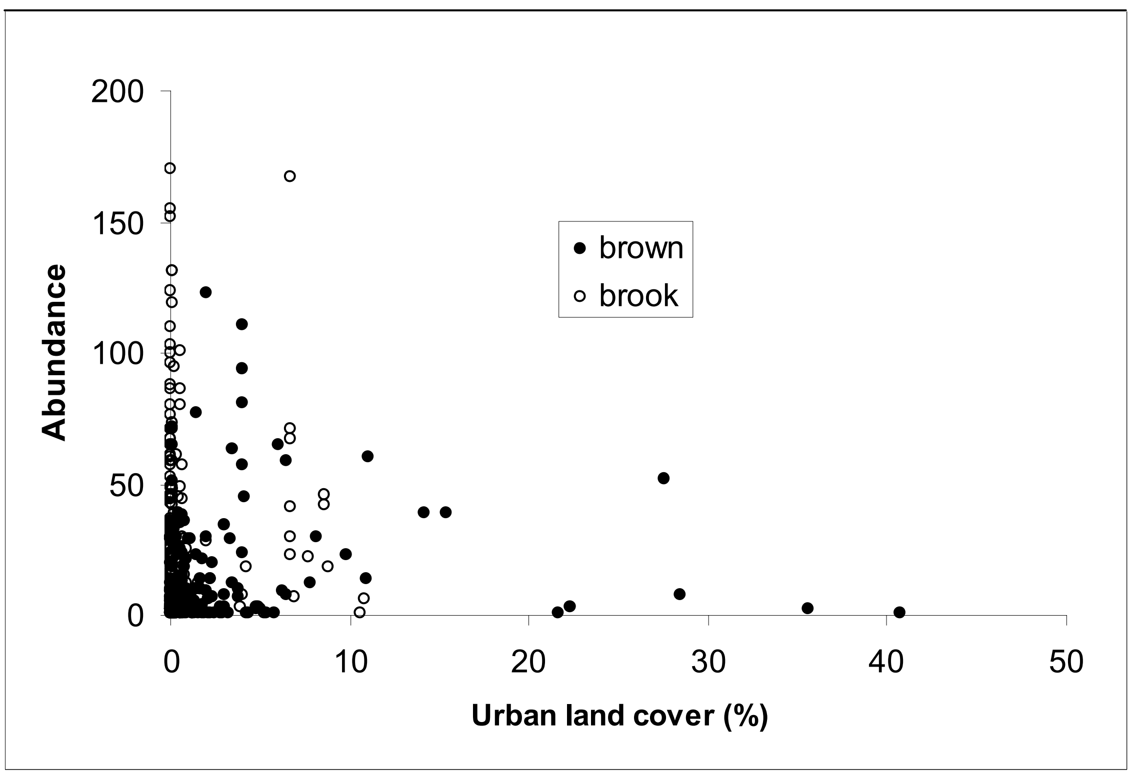


Figure 14-58. Percentage of urban land cover of sites containing brook trout or brown trout for MBSS 1994-2004.

14.6 LAND USE

Streams are affected by the full range of human activities, so it is often impossible to identify all the stressors and causes of degradation. In addition, rivers and streams are hierarchical systems wherein the stressors affecting a local stream may originate elsewhere in the basin. Therefore assessing basin land use is a necessary part of stressor identification. Solutions to stream problems also depend largely on the landscape context. Remediating individual stressors at the site level is of no value if it cannot be sustained within the existing and future land uses in the basin. Maryland and other states recognize that different water quality goals may be necessary where intensive human land uses predominate. Therefore, this section evaluates land use as a measure of anthropogenic influence at the landscape scale.

Basins form natural geographic units for assessing impacts on streams, because land use within the basin (or catchment) upstream of a specific stream site is representative of many of the human activities affecting the stream at that point. As such, land cover serves as a surrogate for a variety of stressors, some of which may be difficult to measure directly. Because no field sampling program will ever be able to visit all parts of all streams throughout the state, the “wall-to-wall” coverage provided by land cover data serves as a useful tool for predicting conditions at sites that cannot be sampled. Geographic information system (GIS) data may be used to develop predictive models linking land cover with instream biological or physical habitat conditions. In evaluating streams across a large area, GIS land cover information can be employed in an initial screening step to locate areas likely to exhibit either desired or degraded conditions and to then target subsequent field sampling to these streams. Depending on management goals, these more detailed investigations can provide the information needed to make informed conservation or restoration decisions.

14.6.1 Maryland Land Uses

Historically, much of Maryland was covered by forest, a sharp contrast to the variety of urban and agricultural uses presently dominating the landscape (Figure 14-59). In Maryland, as in much of the United States, conversions of naturally vegetated lands to urban and agricultural uses have resulted in serious impacts to streams and their inhabitants. Examining land uses as composite stressors, through analyses of relationships with ecological indicators, allows predictions to be made about the extent and severity of ecological impacts associated with varying levels of human use. Urban land produces impervious surfaces, such as roads, parking lots, sidewalks, and rooftops, that cause a rapid increase in the rate at which water is transported from the basin to its stream channels. Effects include more variable stream flows, increased erosion from runoff, habitat degradation caused by channel instability, increased nonpoint source pollutant loading, and elevated temperatures. Urban development of even small portions of a basin (less than 10%) may affect stream biota (Schueler 1994). [Agricultural lands are_strongly associated with high inputs of nutrients and sediment into receiving streams]. The presence of these land uses in the upstream catchment and in the riparian zone both have important, but sometimes differing, effects on stream condition. It is also critical to consider the “legacy” effects of earlier land uses whose effects may still continue long after the land use has changed (Harding et al. 1998). Investigators believe that the sediment loads in many Maryland Piedmont streams were delivered 50-100 years ago (Ray Morgan personal communication).

Associations between urban or agricultural land use and stream biota have been examined in a number of studies (e.g., Klein 1979, Steedman 1988, Richards et al. 1996, Roth et al. 1999). In this section, we report on the relationships observed between land use and several indicators of stream condition for sites sampled by the MBSS from 1995 to 2004, including the fish IBI, benthic macroinvertebrate IBI, and the number and distribution of stream salamanders. Because the MBSS employs a probability-based design, examining land use associations for the sampled sites allows us to make inferences about the effects of land use on biological resources, statewide and within individual basins.

In this section, we specifically examine urban land use, impervious surface, roads, and trash as surrogates for human activities that affect stream quality. In particular, impervious surface is a good surrogate for “flashy” stormwater flows that one-time MBSS sampling cannot capture. Note that the percent coverage by impervious surface for a catchment would be lower than the corresponding value for percent urban land assessed by the MBSS. According to the class definitions used in developing the land cover base data (MRLC 1996 a,b), impervious surfaces make up 30-80% of the low-intensity and 80-100% of high-intensity developed urban land classes. Other land cover classes contribute smaller but possibly significant proportions of impervious surface. Therefore, the values for percent urban land use associated with poor stream quality were expected to be somewhat higher than that for impervious surface effects.

LAND USE AS A BASIN, RIPARIAN, AND LOCAL SCALE STRESSOR ON STREAMS

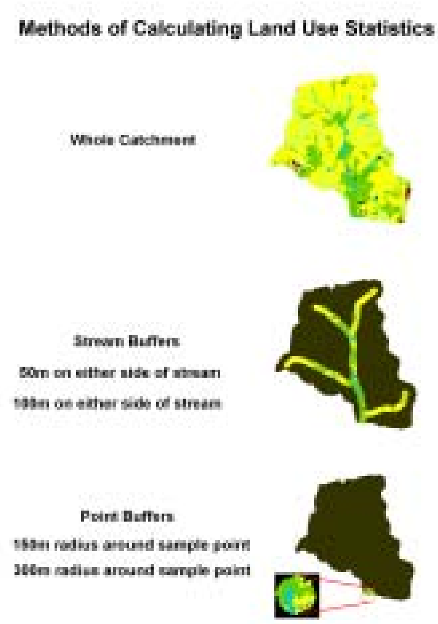
Since the advent of readily available GIS technologies and comprehensive land use coverages, investigators have shown that land use can alter the stream habitat and significantly affect the biotic integrity of streams (e.g., Richards and Host 1994, Wang et al. 1997). Specifically, farming and urbanization in basins are associated with degradation of the invertebrate (Klein 1979, Garie and McIntosh 1986, Jones and Clark 1987) and fish (Scott et al. 1986, Steedman 1988, Roth et al. 1999, Wang et al. 1997, Weaver and Garman 1994) communities in streams and rivers throughout North America. The effects of land use are significant at basin, riparian, and local scales, though the relative influence of land use at each scale varies among studies (Roth et al. 1999, Wang et al. 2001, Vølstad et al. 2003). As an example, the variance in biological indices explained by urban and agricultural land uses at basin and riparian scales in a Wisconsin study were 19% and 35%, respectively, with the interaction accounting for 26% (Wang et al. 2001).

In Maryland, Vølstad et al. (2003) found that urban land use at the catchment scale resulted in the highest coefficient of determination for the regressions of urban land use against fish and benthic IBIs. Land use data for the State of Maryland was extracted from the Federal Region III National Land Cover (NLCD) digital data set (Vogelmann et al. 1998). The percentage of land area in each land use class was calculated at three different spatial scales (see Figure):

- Catchment — includes the entire contiguous basin upstream of a MBSS sample site
- Riparian corridor — the streamside area within a 50-m distance on each side of the streams, for the entire length upstream of a site;
- Local area — area within a 300-m radius circle around a sample site

Of the six landscape classes evaluated, urban land was consistently selected in the one-variable “best” linear regression model (highest), and was included as a variable in all model selections using the maximum method for both fish and benthic IBIs. Linear regression models with one or more additional landscape classes only marginally improved the fit. In the Patapsco basin, the coefficient of determination for a regression of fish IBI against percent urban land use was 0.83 at the catchment scale, as compared with 0.41 at the riparian scale and 0.49 at the local scale; all regressions were highly significant ($p < 0.0001$).

In contrast, the MBSS data did not show a negative relationship between agricultural land use and IBIs, as has been the case in other states (e.g., Gordon and Majumder 2000, Wang et al. 2000). This may be an artifact of the interdependency between percent agricultural land and percent urban land use in the study basins (i.e., when one is low, the other is high). King et al. (2005) used MBSS data in the Coastal Plain of Maryland to investigate the following issues inherent in land use analyses: proportional interdependence , spatial autocor-relation, linkages with abiotic intermediaries, and spatial arrangement. They developed a distance-weighted method for analyzing land use effects to compensate for this spatial autocorrelation.



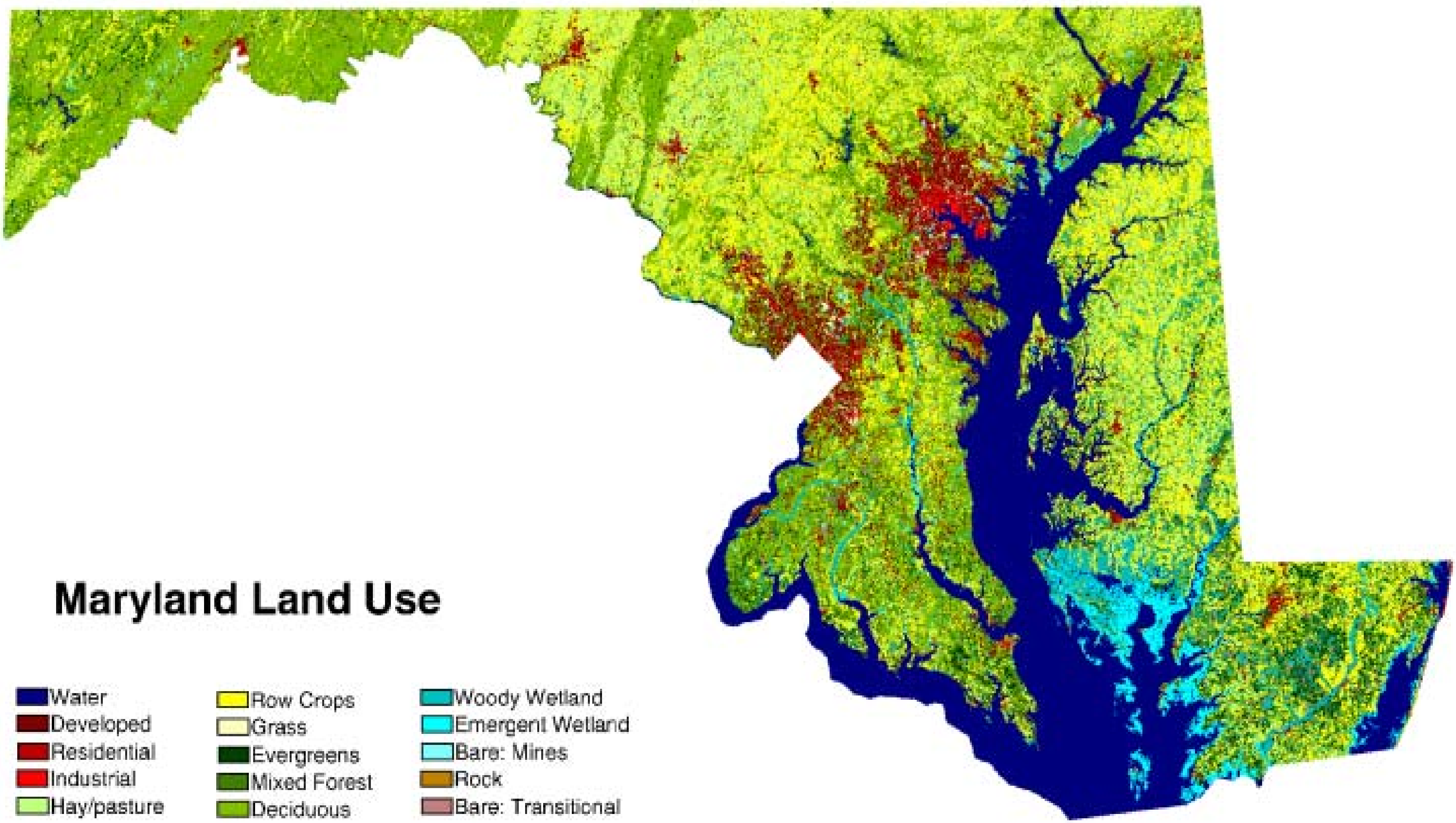


Figure 14-59. Map of land use in Maryland.

A characterization of catchment land use was developed for the basin upstream of each site sampled by the MBSS from 2000 to 2004 using the GIS methods described in Volume 6: Methods. Statewide, the dominant land use in site-specific catchments was forest (mean percent cover of 49%), followed by agriculture (43%) and urban (7%) (Figure 14-60). In individual basins, agricultural land use was greatest at sites in the Upper Eastern Shore basin, with a per-site mean of 67%. Agriculture also dominated in the Upper Western Shore, Ocean Coastal, and the Choptank basins, each with a per-site average of greater than 51%. Sites in the Lower Potomac had a mean of just 22%, while the mean in the remaining basins ranged from 27 to 45% agricultural land. Forest cover was most extensive for sites in the Youghiogheny basin (70%) and least extensive in the Upper Eastern Shore (30%) Patapsco basin (30%). As expected, urban land use was greatest in the Patapsco (23%) and Middle Potomac (21%) basins. Two of the remaining basins: the Lower Western Shore and Patuxent basins contained a mean percentage of urban land use between 10 and 20%. The remaining basins had a mean percentage of urban land use of less than 10%.

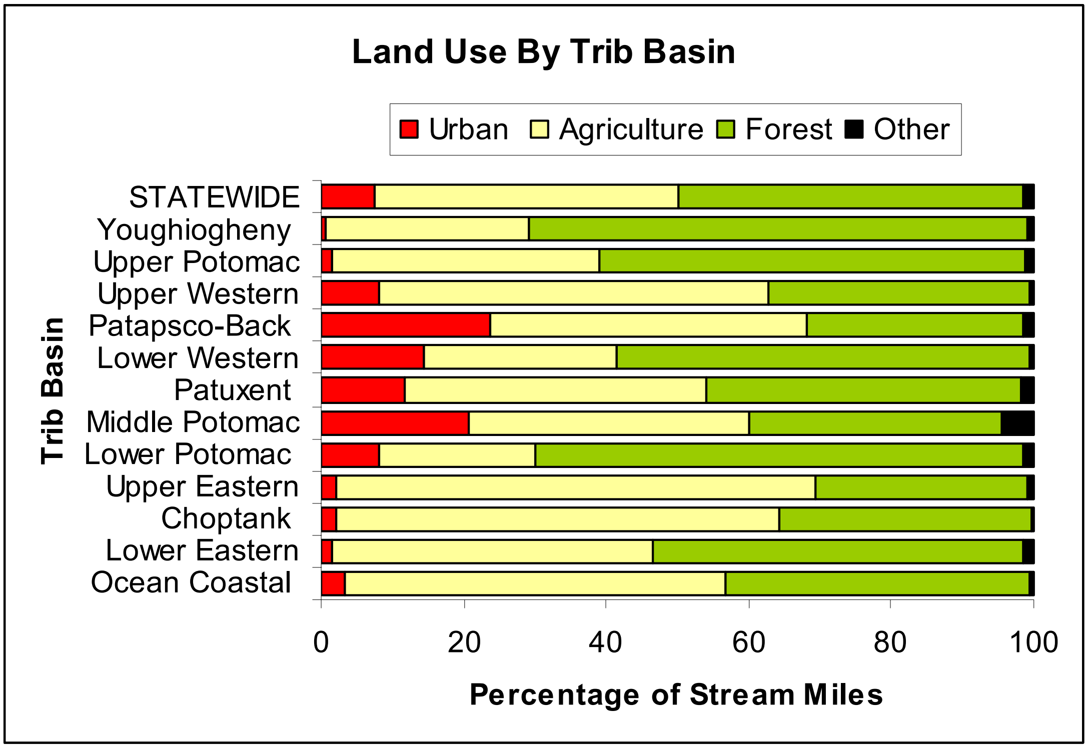
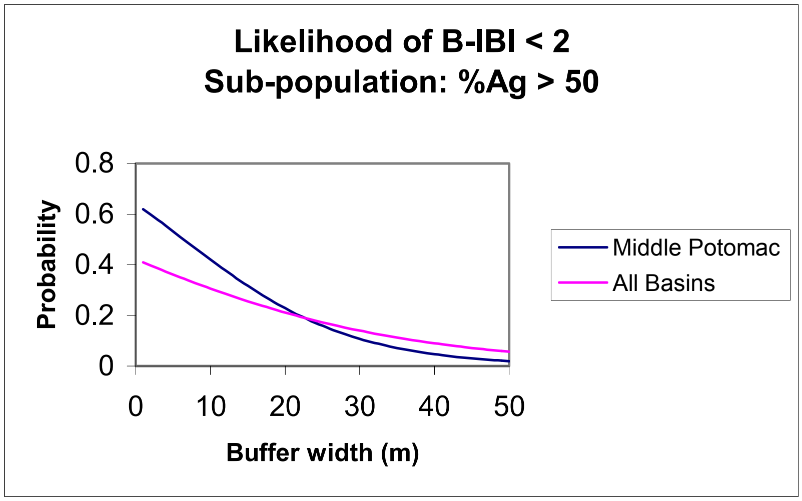


Figure 14-60. Percentage of stream miles by urban, agriculture, forest, and other land use types, statewide and by basins for the 2000-2004 MBSS.

EFFECT OF RIPARIAN BUFFERS ON AGRICULTURAL IMPACTS

Agricultural land uses predominate in much of Maryland, with expected adverse effects on stream condition. The presence of vegetated riparian buffers in agricultural lands varies, so the MBSS looked at the Middle Potomac basin and all Eastern Piedmont basins to determine if the presence of vegetated riparian buffers coincided with high quality streams. We determined that the expected likelihood of very poor condition (benthic index of biotic integrity, B-IBI<2) in catchments with more than 50% agricultural land is reduced from 64% for sites with no riparian buffer, to 11% for sites with 30-m riparian buffer along the streams (see below). This result supports the planting of riparian buffers to potentially mitigate the effects of agricultural land use. In addition to improving the condition of the local stream network and basin, the planting of riparian buffer would benefit the Chesapeake Bay by reducing nutrient loads.



14.6.2 Urbanization

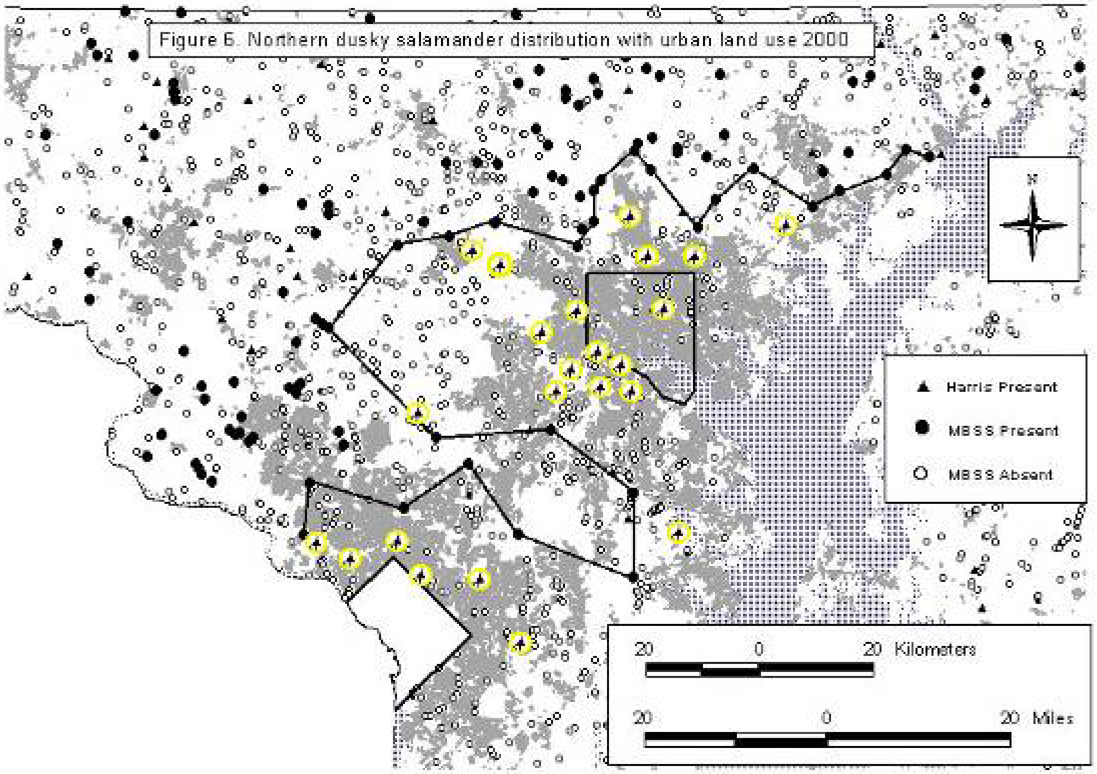
Urbanization is the transformation of rural environments to residential, commercial, or industrial land use, and includes prominent increases in impervious surfaces and roads (discussed below). The amount of land in urban use has accelerated in recent decades to the point that it is the leading cause of water-body impairment affecting more than 130,000 km of U.S. streams and rivers (USEPA 2000a,b,c). Urbanization is also the second-leading cause of species imperilment (next to nonnative species) in the United States (Czech et al. 2000).

Paul and Meyer (2001) describe the many ways that urbanization affects aquatic ecosystems. These include contributing 10,000 times as much fine sediment to streams as do forested basins (Wolman and Schick 1967), carrying higher concentrations of phosphorous and nitrogen than forested or agricultural streams (Osborne and Wiley 1988), and modifying physical habitat through hydrologic changes (Walsh et al. 2004). Although stream channels may ultimately adjust to the altered hydrology, such adjustments may take several decades following urbanization (Henshaw and Booth 2000). Habitat quality in urban streams is often further reduced by active removal of instream woody debris and riparian vegetation.

The physical and chemical changes associated with urbanization strongly influence aquatic biota. Fish and benthic macroinvertebrate communities in urbanized basins commonly exhibit less abundance and lower species diversity (Weaver and Garman 1994, Wang et al. 2000). Anadromous fishes are especially sensitive to urbanization (Limburg and Schmidt 1990). Biological impacts are detectable quite early in the urbanization process. Unlike most agricultural land cover, small amounts of urban land cover, especially near streams, can severely impair biota (Wang et al. 2001). Additional research is needed to determine the relative importance of physical versus chemical effects as the drivers of biological change during urbanization.

Figure 14-61 shows the common inverse relationship between the MBSS benthic IBI and urbanization. MBSS data also show that the percentage of urban land use is the best predictor of a stream failing biocriteria (i.e., being degraded). Specifically, Vølstad et al. (2003) applied logistic regression to quantify how the biotic integrity of streams (using MBSS IBI-based biocriteria) at a local scale is affected by cumulative effects resulting from catchment land uses, point sources, and nearby transmission line rights-of-way (see earlier side bar). Indicators for land use were developed from the remotely

LAND USE CHANGE FRAGMENTS MARYLAND SALAMANDER DISTRIBUTIONS



The advent of readily available land use maps for different time periods (usually from remote sensing) allowed the MBSS to evaluate the effect of urban sprawl occurring in Maryland since the 1960s (Southerland and Stranko, 2005). The map shows the distribution of northern dusky salamander records from both 1960s or earlier (Harris 1975) and the 1995-2004 MBSS in relation to land use in the Baltimore-Washington, D.C. urban corridor. Only presence records (solid triangles) are included from Harris (1975) as no absence records were reported. Both presence (solid circles) and absence (open circles) records are shown for the MBSS. The lack of presence records (and the large number of absence records) in the Baltimore-Washington, D.C. corridor is striking for this and other stream salamander species. Specifically, northern dusky salamander records are conspicuously absent (i.e., southeast of the connected line) in the areas of urban land use (shown as shading in the figure) surrounding both cities. There are substantial regions around the metropolitan areas of Baltimore and Washington, D.C., where long-tailed salamanders and northern red salamanders are also no longer found. Overall, a tally of records from areas that are now urban (based on 2000 land use) for both the 1960s survey and the current MBSS clearly shows that populations of these salamanders have been drastically reduced in urbanizing Maryland.

sensed National Land Cover Data and applied at different scales to three mixed land use drainage basins: Patapsco River, Patuxent River, and Potomac Washington Metro. They determined that the risk of local impairment in nontidal streams rapidly increases with increased urban land use in the catchment area (Figure 14-62). The average likelihood of failing biocriteria doubled with every 10% points increment in urban land, thus an increase in urban land use from 0 to 20% quadruples the risk of impairment. For the basins evaluated in this study, catchments with more than 40–50% urban land use had greater than 80% probability of failing biocriteria, on average. Inclusion of rights-of-way and point sources in the model did not significantly improve the fit for this data set, most likely because of their low numbers. The study indicates that urban land use is the strongest determinant of stream condition in central Maryland. Lastly, in combination with historical distribution information, MBSS sampling data illustrate the dramatic effect of urbanization on a stream salamander (see side bar).

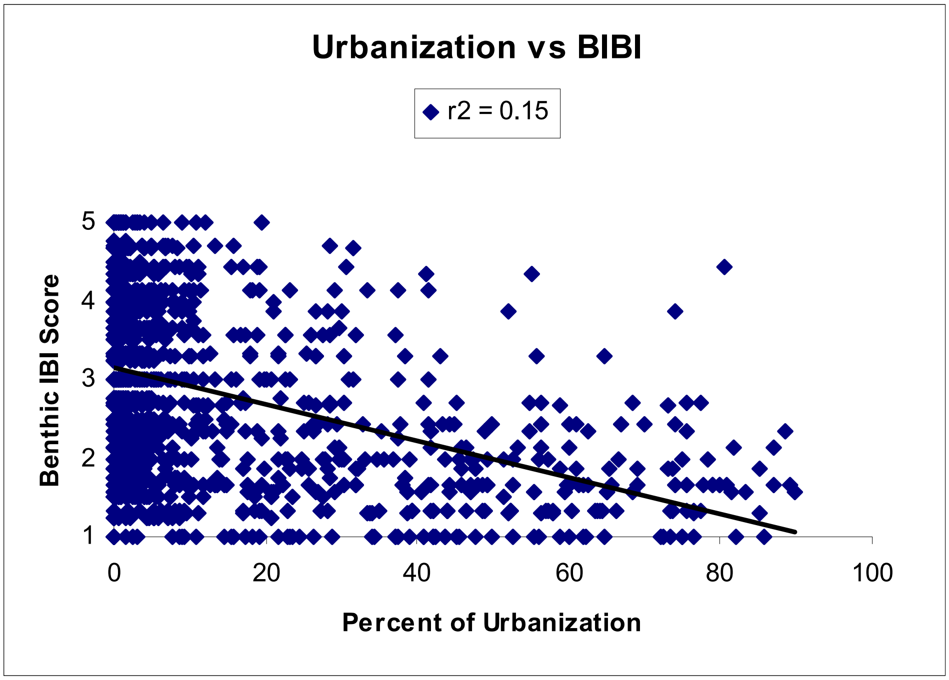


Figure 14-61. Relationship between percentage of urban land use and the benthic-macroinvertebrate index of biotic integrity, statewide for the 2000-2004 MBSS.

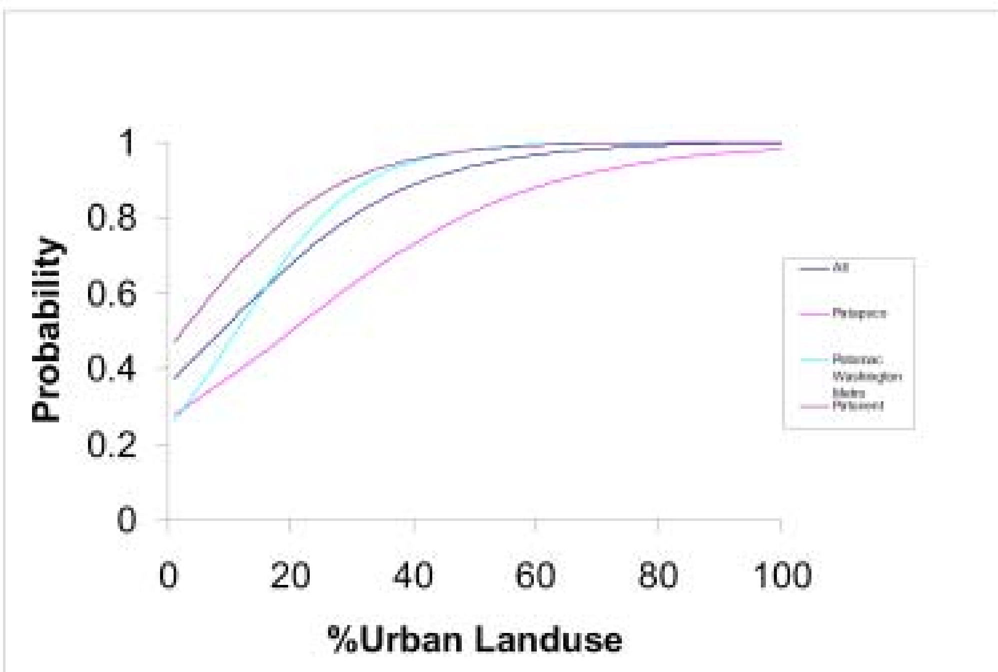


Figure 14-62. The probability of failing the Maryland biocriteria vs. percentage of land use in the catchments of MBSS sample sites for three central Maryland basins and across these basins.

14.6.3 Impervious Surfaces

The amount of imperviousness in a basin is the total contribution of paved surfaces, roofs, other solid structures, and less pervious soils found in sports fields and other hard packed areas. While imperviousness is highest in urban areas, it exists to varying degrees in suburban and rural environments as well. Roads are often the biggest contributor to impervious area (May et al. 1997). The proliferation of impervious surfaces fundamentally alters the timing of precipitation runoff, resulting in higher peak flows during storms and lower base flows (e.g., Wang et al. 2001, Walsh et al. 2004). During storm events, water on impervious surfaces is routed more quickly to the stream, resulting in current velocities unsuitable for many organisms. The energy associated with high flows also results in greater scouring and movement of bedload, increasing mortality of less mobile species. The extra energy associated with high flows may also precipitate channel incision through accelerated downcutting (Booth 1990). When downcutting occurs, the stream becomes less connected to its floodplain and streambank vegetation is less able to protect against bank erosion. When the energy of the stream is focused laterally, channel widening occurs, resulting in an increase in the width-to-depth ratio and a reduction in habitat quality for many species. During dry periods, the less water percolating into the soil during storms results in reductions in baseflow. This reduction further exacerbates the shallowing of habitat and may markedly slow current velocity. Consequently, urban streams tend to have wide, silty channels with relatively little water.

Higher flows during storms also more readily transport sediment, nutrient-laden surface runoff, toxic contaminants, large woody debris, rootwads, Coarse Fine Particulate Organic Matter (CPOM), Fine Particulate Organic Matter (FPOM), and Dissolved Organic Matter (DOM) downstream. These flows also result in decreased nutrient spiraling, increased turbidity/siltation, reduced amounts of habitat refugia, and potentially lethal contaminant concentrations. In urban areas such as Baltimore City that feature combined storm and sewer drains, high flow events result in elevated bacterial and nutrient levels, including potentially lethal concentrations of ammonia. When high flow events occur after extended periods of dry weather, a “first flush” of polycyclic aromatic hydrocarbons (PAHs) can kill many organisms. When high flow events occur during hot summer conditions, the heated water running off hot pavement and rooftops can result in unlivable stream temperatures. During and after winter storm events, concentrations of chlorides and heavy metals can far exceed tolerance limits for freshwater biota. In total, increased imperviousness from urbanization causes numerous deleterious changes to stream habitats, often resulting in severely impaired biological communities.

Tolerant macroinvertebrate and fish species quickly replace sensitive species as impervious surfaces cover 5 to 15% of a basin's area (Scheuler 1994, Klein 1979). However, Roth et al. (1999) found that significant loss of fish, benthic macroinvertebrate and aquatic herpetofauna species richness occurred at levels below this threshold. Biotic communities often change little after impervious land cover exceeds 20% of a basin (Booth and Jackson 1997; Wang et al. 2000; but see Morley and Karr 2002 for a biotic response when impervious land cover exceeds 50%).

As with urban land use, both fish and benthic IBIs decrease with increased impervious surface. Figures 14-63 and 14-64 illustrate poorer biological condition as the percentage of impervious surface exceed 5% and 20%. Figures 14-65 shows a similar decreasing relationship with the number of stream salamander species.

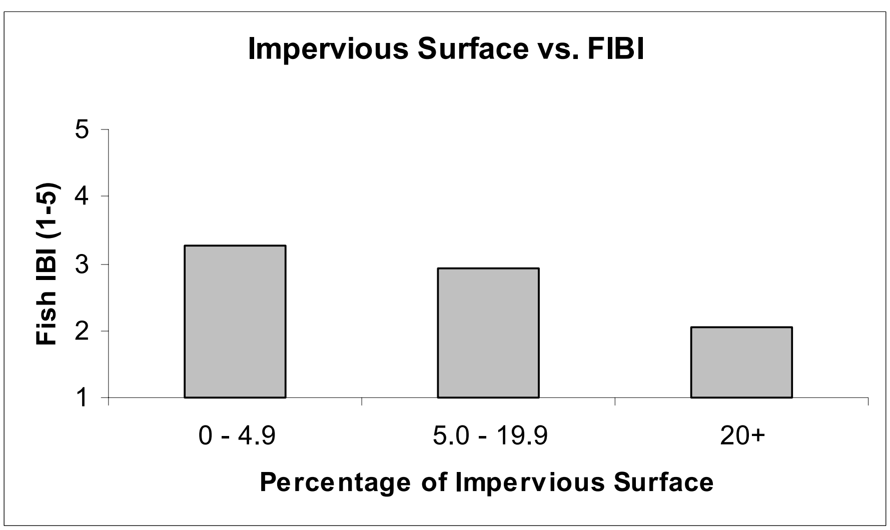


Figure 14-63. Relationship between impervious surface and the fish index of biotic integrity score (IBI), statewide for the 1994-2004 MBSS.

MAXIMUM IMPERVIOUS SURFACES TOLERATED BY FISH AND STREAM SALAMANDERS IN MARYLAND

Impervious land cover maximums for fish and stream salamander species collected by MBSS during 1994-2004. Maximums were not reported for estuarine, big river species, and species that occurred at less than 10 sites.

Common Name	Scientific Name	Impervious Maximum
***johnny darter	*Etheostoma nigrum*	0.1
***seal salamander	*Desmognathus monticola*	0.2
***mottled sculpin	*Cottus bairdi*	0.6
***Allegheny mountain dusky salamander	*Desmognathus ochrophaeus*	1.4
northern spring salamander	*Gyrinophilus p. porphyriticus*	1.9
checkered sculpin	*Cottus sp.*	2.0
mud sunfish	*Acantharchus pomotis*	2.5
***rainbow darter	*Etheostoma caeruleum*	2.6
swamp darter	*Etheostoma fusiforme*	3.0
brook trout	*Salvelinus fontinalis*	4.2
long-tailed salamander	*Eurycea longicauda*	4.8
rosyface shiner	*Notropis rubellus*	5.5
shield darter	*Percina peltata*	5.8
longear sunfish	*Lepomis megalotis*	7.1
warmouth	*Lepomis gulosus*	7.2
comely shiner	*Notropis amoenus*	8.7
banded sunfish	*Enneacanthus obesus*	9.2
river chub	*Nocomis micropogon*	11.5
glassy darter	*Etheostoma vitreum*	12.0
flier	*Centrarchus macropterus*	12.5
pirate perch	*Aphredoderus sayanus*	12.5
tadpole madtom	*Noturus gyrinus*	12.5
American brook lamprey	*Lampetra appendix*	12.9
blue ridge sculpin	*Cottus caeruleomentum*	13.7
*brown trout	*Salmo trutta*	14.3
greenside darter	*Etheostoma blennioides*	15.3
sea lamprey	*Petromyzon marinus*	15.3
spotfin shiner	*Cyprinella spiloptera*	15.3
fallfish	*Semotilus corporalis*	15.5
northern red salamander	*Pseudotriton ruber*	15.6
Potomac sculpin	*Cottus girardi*	16.1
margined madtom	*Noturus insignis*	16.1
northern hogsucker	*Hypentelium nigricans*	16.1
cutlips minnow	*Exoglossum maxillingua*	16.9
fantail darter	*Etheostoma flabellare*	20.2
northern dusky salamander	*Desmognathus fuscus*	20.2
silverjaw minnow	*Notropis buccatus*	20.2
**black crappie	*Pomoxis nigromaculatus*	22.1
rock bass	*Ambloplites rupestris*	22.1
bluntnose minnow	*Pimephales notatus*	24.3
**largemouth bass	*Micropterus salmoides*	24.3
smallmouth bass	*Micropterus dolomieu*	24.3
yellow bullhead	*Ameiurus natalis*	24.3
eastern silvery minnow	*Hybognathus regius*	24.5
mosquitofish	*Gambusia holbrooki*	24.5
pearl dace	*Margariscus margarita*	25.7
central stoneroller	*Campostoma anomalum*	26.7
spottail shiner	*Notropis hudsonius*	26.8
bluespotted sunfish	*Enneacanthus gloriosus*	27.1
chain pickerel	*Esox niger*	27.1
least brook lamprey	*Lampetra aepyptera*	27.1
redfin pickerel	*Esox americanus*	27.1
*rainbow trout	*Oncorhynchus mykiss*	27.5
satinfin shiner	*Cyprinella analostana*	28.0
swallowtail shiner	*Notropis procne*	28.0
bluegill	*Lepomis macrochirus*	28.9

Common Name	Scientific Name	Impervious Maximum
common shiner	*Luxilus cornutus*	28.9
green sunfish	*Lepomis cyanellus*	29.6
American eel	*Anguilla rostrata*	30.6
longnose dace	*Rhinichthys cataractae*	30.6
white sucker	*Catostomus commersoni*	30.6
tessellated darter	*Etheostoma olmstedi*	30.7
eastern mudminnow	*Umbra pygmaea*	31.6
golden shiner	*Notemigonus crysoleucas*	31.6
pumpkinseed	*Lepomis gibbosus*	31.6
blacknose dace	*Rhinichthys atratulus*	37.6
creek chub	*Semotilus atromaculatus*	37.6
creek chubsucker	*Erimyzon oblongus*	37.6
fathead minnow	*Pimephales promelas*	37.6
northern two-lined salamander	*Eurycea bislineata*	37.6
redbreast sunfish	*Lepomis auritus*	37.6
rosyside dace	*Clinostomus funduloides*	37.6
brown bullhead	*Ameiurus nebulosus*	41.2
goldfish	*Carassius auratus*	41.2

* Maximum tolerance is probably not this high because species is anually stocked into streams and may not be a long term resident

** Maximum tolerance is probably not this high because species is often displaced into streams from ponds and may not be a long term resident

***Maximum tolerance may not be this low because species is restricted to small portions of Maryland with little urbanization

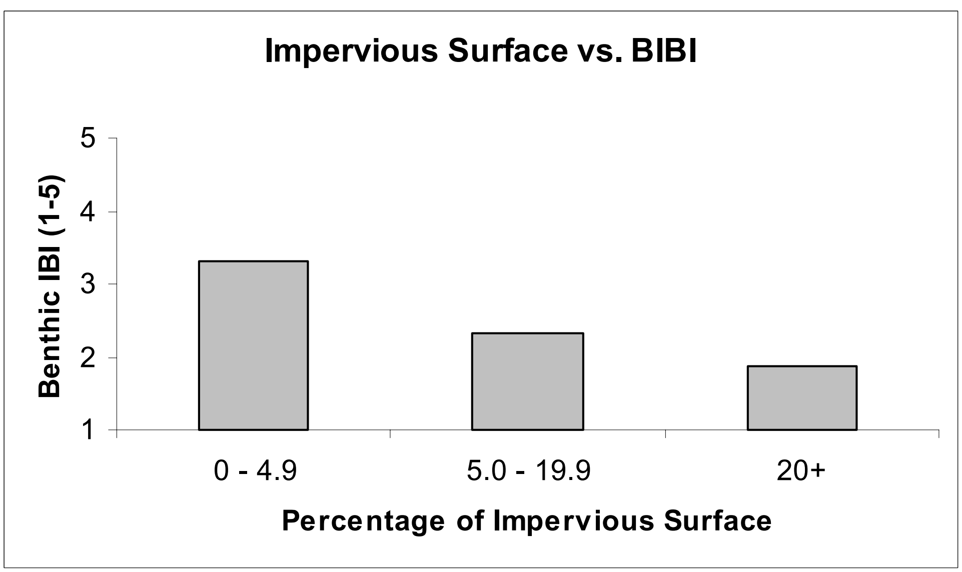


Figure 14-64. Relationship between impervious surface and the benthic-macroinvertebrate index of biotic integrity score (IBI), statewide for the 1994-2004 MBSS.

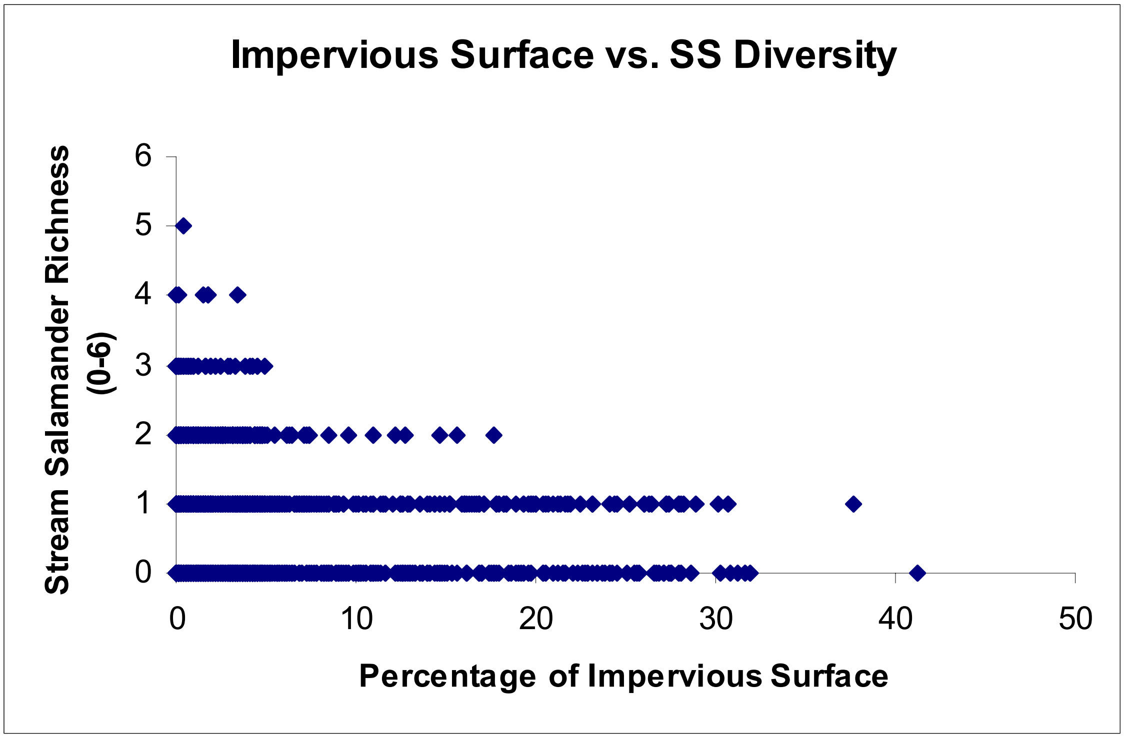


Figure 14-65. Relationship between impervious surface and the number of stream salamanders, statewide for the 1994-2004 MBSS.

14.6.4 Roads

While impervious surfaces are an excellent surrogate for hydrological changes affecting streams, roads may be the best surrogate of human activities across the entire landscape (Southerland 1995, Spellerberg 2002). The ecological effects of roads extend 100 to 1,000 m (average of 300 m) on each side of four-lane roads (Forman and Deblinger 2000). These effects, which stem from both construction and use, vary considerably in type and degree among regions and particular roads. Effects on biological populations and communities, as well as key ecological processes, can be dramatic and probably contribute to local extirpation and regional endangerment of many fishes (Angermeier et al. 2004).

The primary impacts of road construction are direct alteration of the stream channel and indirect acceleration of fine sediment loading from exposed soils. As described above, these alterations can change channel depth, pool-to-riffle ratio, percent fines in substrates, and cover availability. Once constructed, roads still have major effects on water quality (e.g., via toxic spills and runoff), habitat quality (via sediment loading and channel modification), and habitat connectivity (via barriers to movement). Many road crossings over streams constrain movements by small fishes that may be essential for individuals to complete their life cycles and for metapopulations to remain viable (Angermeier et al. 2004). To the extent that roads continue to contribute fine-sediment loading after construction, aquatic biota suffer (Waters 1995, Wood and Armitage 1997). Several studies have shown elevated concentrations of contaminants in aquatic animals near roads (Van Hassel et al. 1980, Stemberger and Chen 1998). Roads also provide human access to streams and can enhance the spread of non-native fishes, mollusks, and pathogens (Trombulak and Frissell 2000).

In Maryland, as in many other parts of the Country, vehicle miles traveled have increased in recent decades (Figure 14-66). This is a result of greater dispersed development (urban sprawl) and our automobile-based lifestyle. As the number of vehicle miles traveled increases, the road infrastructure grows and the impact of each segment (supporting more traffic) increase. Figure 14-67 shows the distribution of road crossings in Maryland.

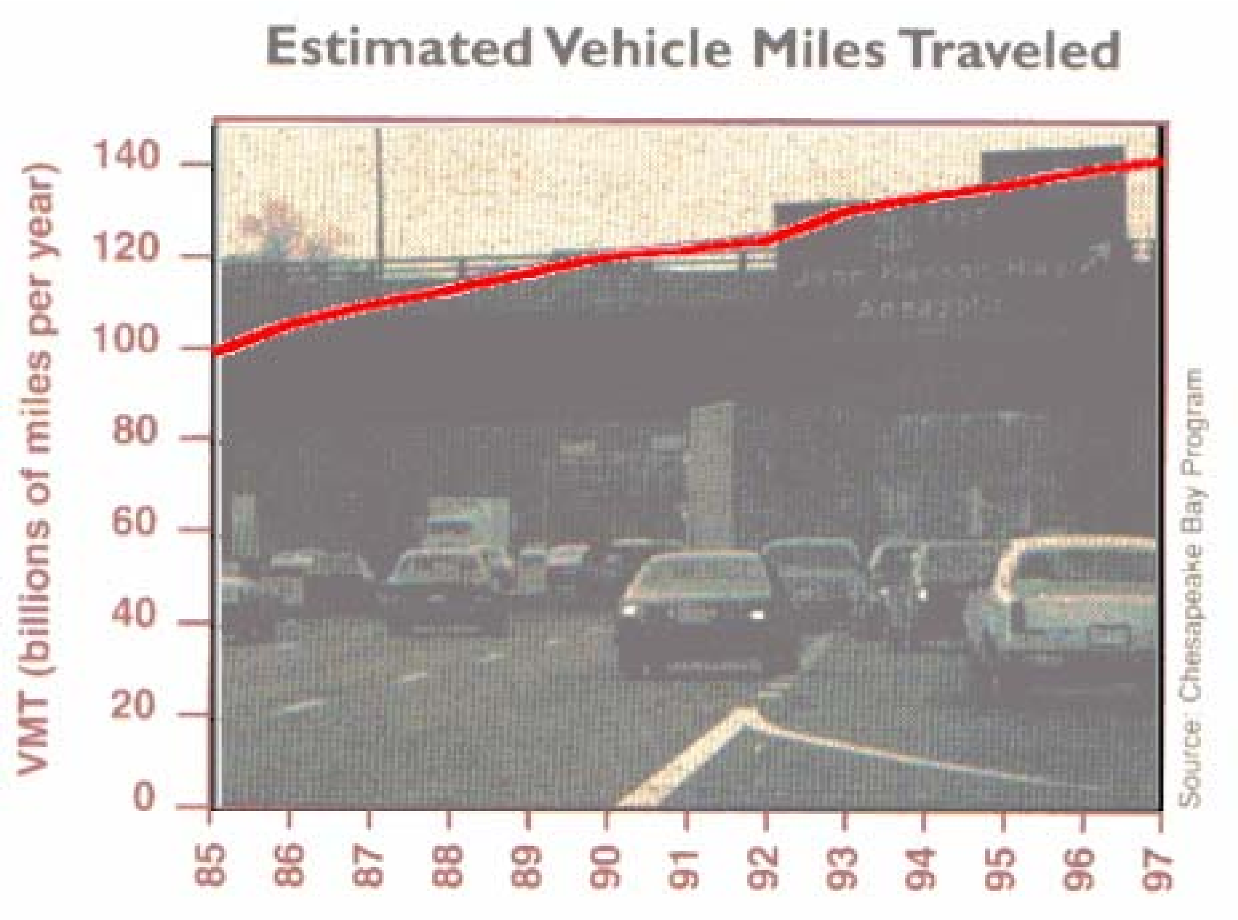


Figure 14-66. Vehicle miles traveled (VMT) in billions per year throughout the Chesapeake Bay basin.

14.6.5 Trash

Another useful surrogate for human activities at the scale of the MBSS site is the amount of trash. Originally called "aesthetics," this "absence of trash" score is based on a 0 to 20 scale that increases as the amount of trash decreases. A score of 20 is the complete absence of trash or other obvious evidence of human presence. Figures 14-68 and 14-69 show the increasing relationships between the absence of trash and benthic IBI and PHI, respectively.

14.7 RELATIVE RISKS AND CUMULATIVE IMPACTS OF STRESSORS ON MARYLAND STREAMS

Identifying stressors is critical to the development of management actions by the State and others to restore or protect the desired condition of streams. In particular, stressor identification is critical to implementing Total Maximum Daily Loads (TMDLs) developed by MDE to address streams impaired under CWA Section 303(d). Stressor information is also critical for active restoration programs for Maryland's streams such as Maryland DNR Watershed Restoration Action Strategies (WRAS) and Chesapeake Bay Program Tributary Strategies.

Identifying stressors, however, is not sufficient to guide effective management actions for Maryland streams. The State, Counties, and other natural resource stewards must assess the relative risks posed by different stressors at site, basin, and regional scales. Only by comparing these risks can effective stream restoration or protection be attained. Comparative risk assessment is an emerging discipline (U.S. Environmental Defense 2004) that tries to answer the question—Is one environmental problem worse than another? The search for this answer provides a good forum for discussing environmental issues. It also leads to better environmental management by weeding out obsolete programs, promoting efficiency, establishing credibility, improving relationships among stakeholders, and increasing awareness of the environment by the government and the public. Critical to comparative risk discussions is the understanding that better environmental conditions provide a better quality of life, and that preservation and protection are always more cost-effective than restoration. Determining the comparative risks facing Maryland streams involves assessing not only the severity, extent, and reversibility of risks, but

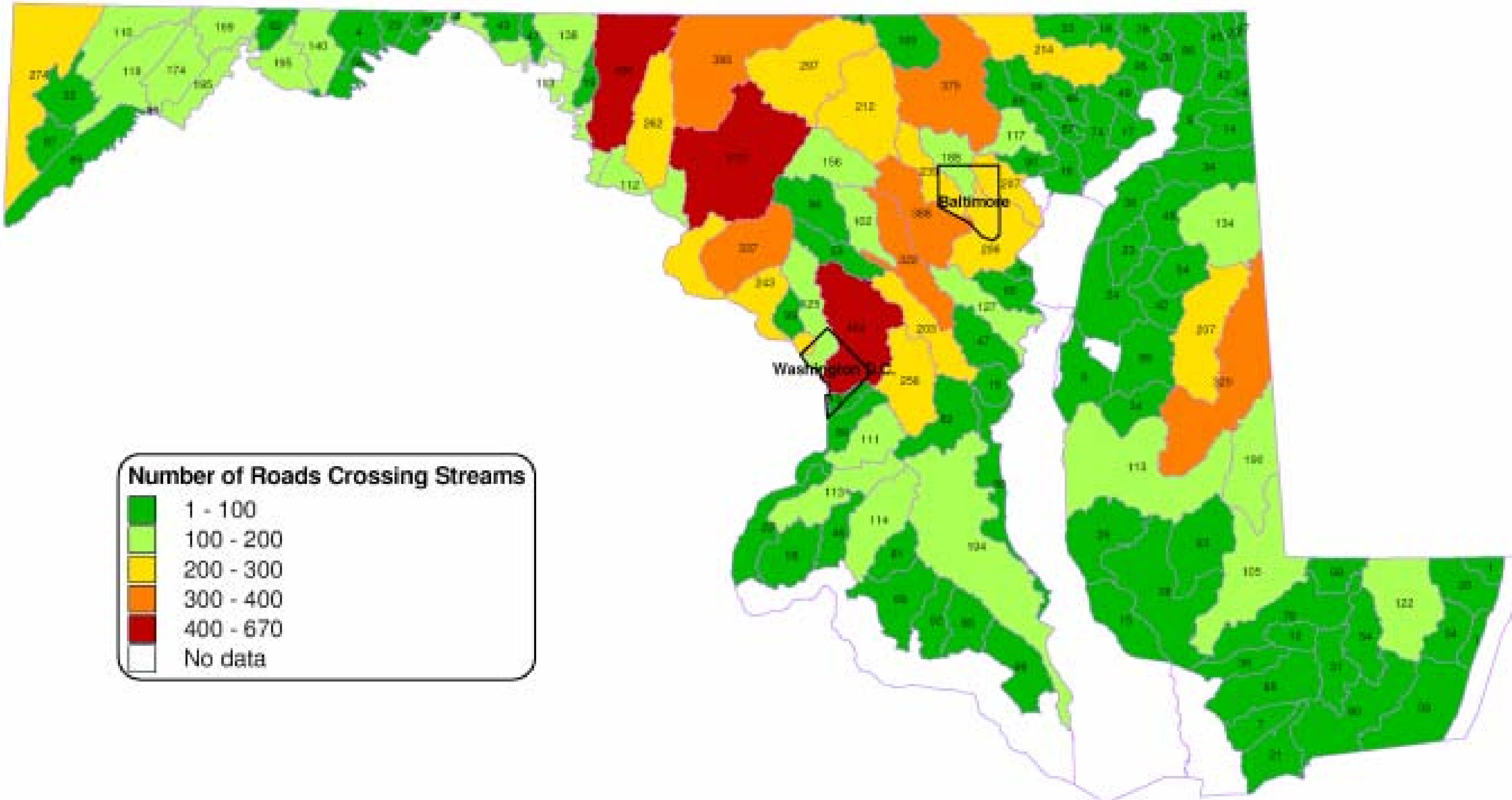


Figure 14-67. Number of road crossings of streams by basin in Maryland.

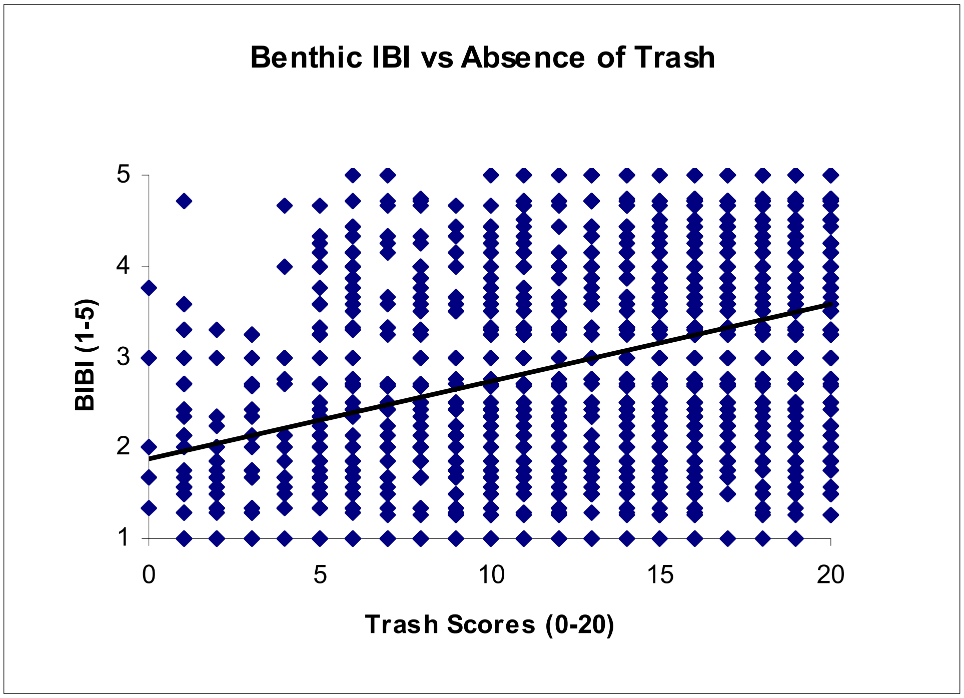


Figure 14-68. Relationship between the benthic index of biotic integrity (IBI) and the absence of trash score, statewide for the 1995-2004 MBSS.

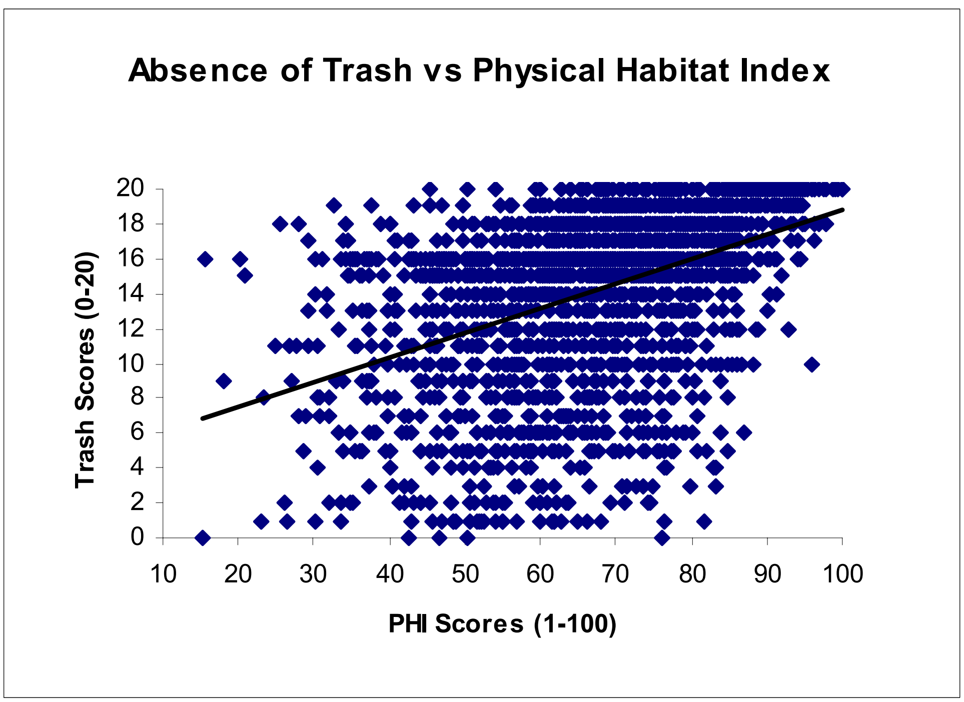


Figure 14-69. Relationship between the physical habitat index (PHI) and the absence of trash score, statewide for the 1995-2004 MBSS.

also the uncertainty associated with these risks. Ultimately, a comparative risk assessment will lead to a strategic plan for stream protection and restoration that includes priorities for action, resource needs and potential sources of funding, and a means of tracking and documenting success.

Comparisons of relative risks must be accompanied by assessments of the cumulative impacts that result. It is the total effect of stressors combined in space and time that degrade stream conditions. Odum (1982) aptly described degradation from cumulative impacts as “the tyranny of small decisions.” Atmospheric deposition is a good example of cumulative impacts—natural resources in a stream become degraded when the total loading of acid exceeds the threshold assimilative capacity of the receiving basin. Complicating cumulative effects analysis for streams is the fact that streams and rivers are hierarchical systems wherein the character of a local stream site is affected by the larger stream network and basin to which it belongs. This means that to fully understand the multiple, cumulative impacts on stream systems, conditions at a broad landscape scale, as well as the local or site-specific scale, must be assessed. Equally important is the recognition that solutions to stream problems must consider the cumulative (and sometime synergistic effects) of multiple stressors. For example, while water chemistry results may indicate that acidic deposition is the likely cause of degraded fish communities at a particular site, there may be other stressors on that stream that would continue to inhibit fish even if the acidification was ameliorated. Loss of biota where riparian vegetation has been removed may be caused by hydrological changes (i.e., that accelerate bank erosion and sedimentation) or simply the loss of allochthonous inputs (e.g., leaf fall). In other cases, refugia within a local stream network may mitigate severe episodic stresses. This illustrates the need to include landscape-level information in the ecological assessment process. Only by using an integrated multiple-scale and multiple-stressor approach can the cumulative effects on streams be assessed and ameliorated.

Table 14-15 is an example from the first attempt to quantify relative risks posed by 22 stressors to Maryland streams, including some that are not monitored by the MBSS (see Volume 12: Stream and Riverine Biodiversity). This table includes a qualitative evaluation of the severity, extent, reversibility, trends, and degree of understanding (uncertainty) of each stressor. Refinement of this list of relative risks will occur as more MBSS analyses are conducted and as additional data from other sources are incorporated.

14.7.1 Most Severe Stressors Affecting Maryland Streams

Determining the relative risks posed by stressors is important at the scale of management actions. Relative risk includes two components: the importance (or severity) and the prevalence (or extent) of each stressor. We present the severity of ten stressors to Maryland streams as the probability of poor fish and benthic IBI scores given stressor scores above the threshold for degradation, divided by the probability of poor IBI scores given stressor scores below the threshold for degradation. Risk severity scores > 1 indicate which stressors have the strongest effect on streams statewide (Figure 14-70). For fish, acid mine drainage is the most severe stressor; for benthic macroinvertebrates, low dissolved oxygen is the most severe. Urban land use is the most severe stressor overall.

The analysis includes the following ten stressors (shown with the threshold values indicating degradation risk):

- Urban > 5%
- No Riparian Buffer
- Channelization
- NO3 > 5 mg/l
- DO < 3 mg/l
- Acid Deposition Present
- Acid Mine Drainage Present
- Bank Stability Sub-optimal or Poor (<= 11)
- Invasive Plants Present
- Invasive Fish or Mussels Present

These ten stressors are meant to be a representative but incomplete list. The thresholds of concern for each stressor were selected based on expert consensus and analyses to date on the MBSS data. In particular, stressor values that result in demonstratively lower fish or benthic IBI scores have been used as thresholds. Additional stressor analyses are being conducted with the MBSS data and thresholds may be revised in the future.

14.7.2 Extent of Major Stressors Affecting Maryland Streams

While acid mine drainage is a severe stressor, it has the smallest extent statewide (i.e., number of stream miles affected) of the ten major stressors (Figure 14-71). The presence of invasive plants and animals are the most extensive stressors statewide, at 85% and 52% of stream miles, respectively. Poor bank stability affects 25% of stream miles and acid deposition affect 21%, compared to 1% for acid mine drainage. Figure 14-72 shows the extent of each of these major stressors as the percentage of stream miles by each Maryland county with stressor values exceeding the threshold for degradation.

Table 14-15. Relative risks posed to an example Maryland basin (note that relative risks for all Maryland basins are in Volume 9: Stream and Riverine Biodiversity).

Basin: Gwynns Falls

Category	Subcategory	Name	Extent	Trend	Severity	Persistence	Reversibility
Chemical	Non-point Source	Organic Matter Retention	2	3	3	4	3
Chemical	Non-point Source	Acid Deposition/ Low pH	0	2	4	4	2
Chemical	Non-point Source	Acid Mine Drainage	0	1	5	5	1
Chemical	Non-point Source	Excess Nitrates	3	3	2	3	3
Chemical	Non-point Source	Excess Phosphorus	3	3	2	3	4
Chemical	Non-point Source	Mercury Deposition	2	3	2	4	2
Chemical	Point Source	Pathogens/Endocrine disruptors	3	3	4	2	3
Chemical	Point Source	Industrial (NPDES)	4	2	3	3	3
Chemical	Point Source	Agricultural Pesticides	2	2	2	3	3
Chemical	Point Source	Dissolved Oxygen	0	3	4	2	3
Future Changes		Land Conversion	1	3	4	5	2
Future Changes		Sea Level Rise	0	3	5	5	1
Habitat Alteration		Wetland Loss	2	3	3	4	2
Habitat Alteration		Channelization	4	2	3	3	3
Habitat Alteration		Forest Fragmentation	5	3	2	3	4
Habitat Alteration		Ground Water withdrawal	1	2	2	2	4
Habitat Alteration		Migration Barriers	3	2	3	3	2
Habitat Alteration		Runoff/Baseflow/Down Cutting	5	3	3	4	2
Habitat Alteration		Sedimentation	2	3	3	4	3
Habitat Alteration		Surface Water Withdrawal	4	2	2	2	4
Non-natives		Non-native Species (aquatic)	4	2	2	3	2
Non-natives		Invasive Plants (riparian)	4	3	2	3	2

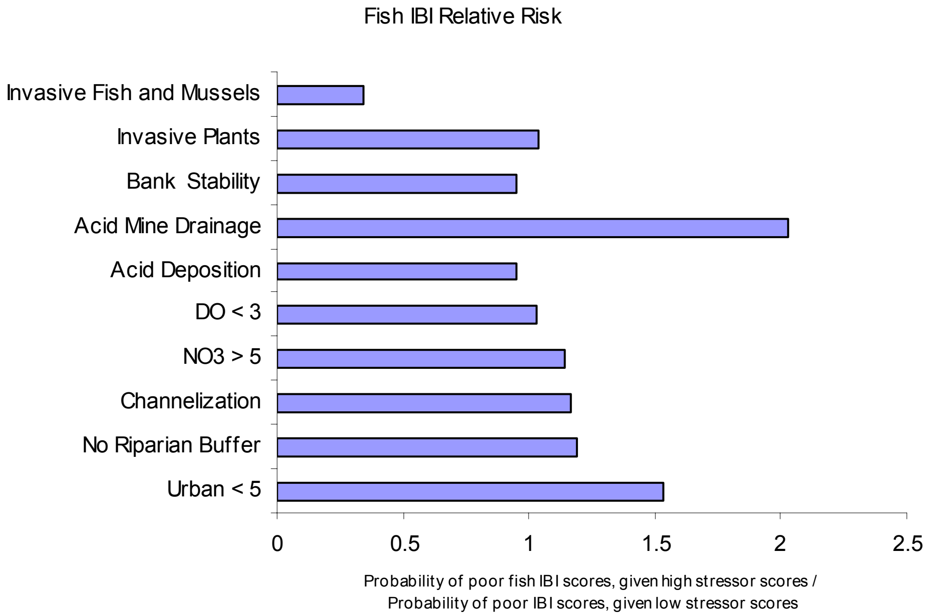
EXTENT (0-5) Based on the estimated percentage of stream miles or, in some cases, areas in the basin that are affected
0 = None or negligible
1 = 1-10%
2 = 11-20%
3 = 21-30%
4 = 31- 60%
5 = 61-100%

TREND (1-5) Based on the projected rate of change and immediacy of the impact
0 = Threat extent decreasing over time, either due to human intervention or natural rejuvenation
1 = Threat extent unchanging
2 = Threat slowly getting worse; up to 0.25% change per year
3 = Threat extent getting worse; up to 0.5% change per year
4 = Threat extent steadily growing, up to 2% change per year
5 = Threat extent rapidly growing, 2 or more percent change per year

SEVERITY (0-5) Based on the estimated or known impact to aquatic ecosystems
0 = No impact likely
1 = Mild
2 = Moderate; degradation of some forms of biological function; detectable shift in community structure and species loss
3 = Serious; significant loss of biological function, communities often dominated by tolerant generalists and/or richness declines
4 = Very serious; heavy loss of biological function; only tolerant species remain
5 = Catastrophic; near-total loss of biological function in affected areas

PERSISTANCE (1-5) Based on duration of impact
0 = Recovery nearly immediate
1 = Short duration, substantial recovery possible in less than 1 year
2 = Moderate duration, substantial recovery possible within 5 years
3 = Long duration, substantial recovery possible within 5-50 years with human remediation
4 = Extreme duration, substantial recovery not likely for 50 to 100s of years, even with intensive human intervention
5 = Essentially permanent environmental feature lasting hundreds of years, even with intensive human intervention

REVERSABILITY (1-5) Based on the degree of difficulty to reduce or eliminate the threat
1 = Only correctable using extreme or unproven measures and at extreme relative cost
2 = Mostly correctable, but at very high socioeconomic cost
3 = Correctable using existing technology, but at high relative cost (social or economic)
4 = Correctable with existing technology and moderate cost
5 = Readily remedied using existing technology



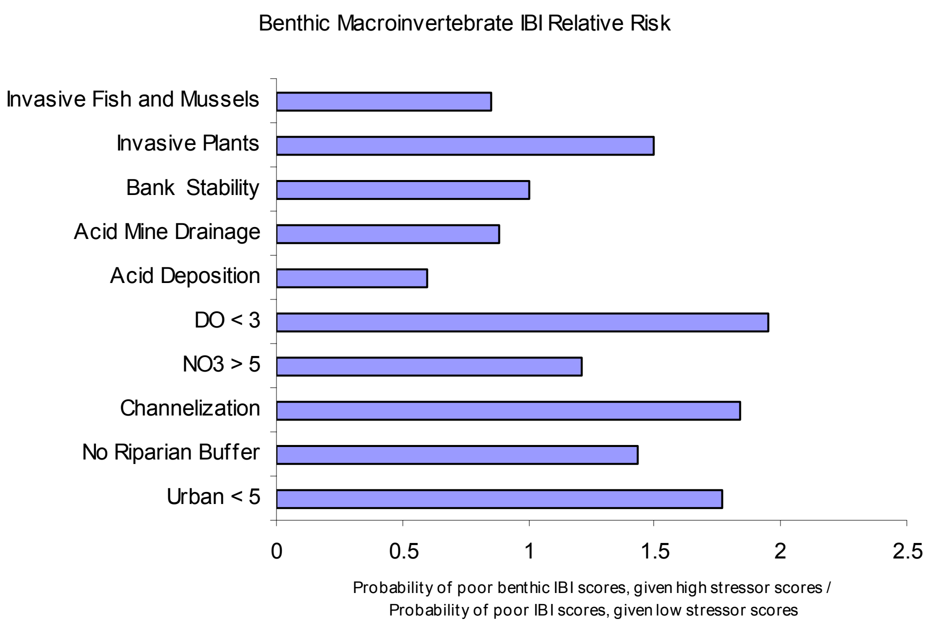


Figure 14-70. Importance of ten stressors to FIBI and BIBI scores.

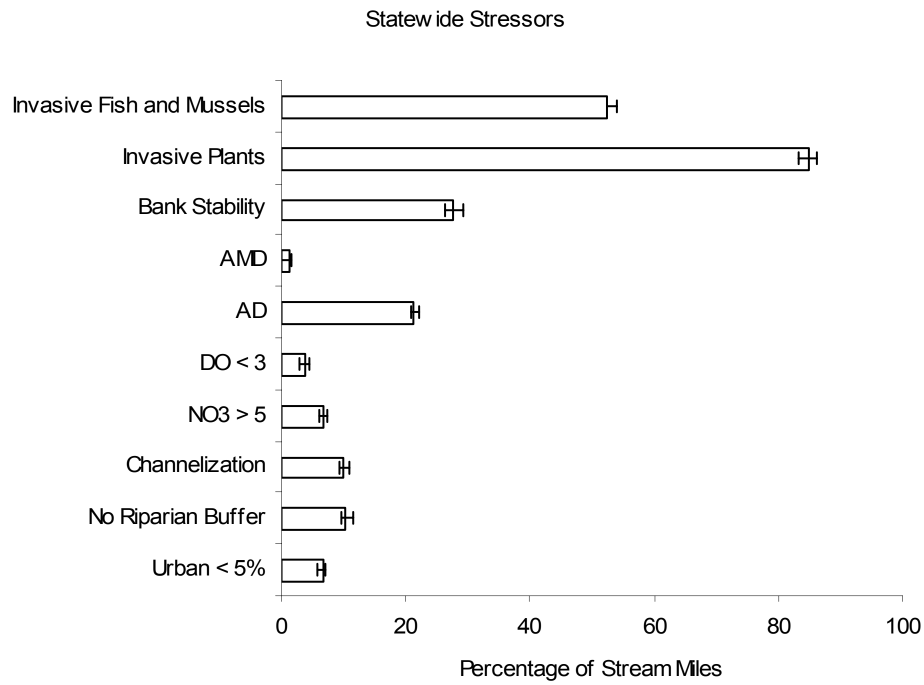


Figure 14-71. Statewide stressors extent for MBSS 2000-2004 data.

14.7.3 Stressor Identified by Loss of Fish Species

The combined effect of stressors can also be determined by identifying where individual fish species are lost. Table 14-16 lists the major stressors to fishes in Maryland streams based on the loss of fish species as determined by the Prediction and Diagnosis Model of Stranko et al. (2005). This model predicts species that should be present in the absence of major anthropogenic stress using variables that do not typically change with human influences (e.g., stream size, geology, altitude, gradient, drainage basin where species occurs). Conservative predictions were generated by only predicting that species should occur where conditions for all predictor variables are optimal. Where species were predicted to occur but were absent, stressors were identified. A stressor to a species was any value outside the tolerance thresholds for the species. The methods used apply this model are described in section 14.8.

In this analysis, a total of 398 MBSS sites had some stressor identified. This was 45% of the total sites the model was applied to and 51% of the sites with at least one predicted species absent. Exactly 71 sites had urban or impervious identified as a stressor; that is 8% of the 889 total sites used in the model and 18% of the 398 where stressors were identified.

Table 14-16 shows the percentage of stream miles where species were absent due to stressors. Statewide results indicate that physical habitat alterations are the most pervasive stressors responsible for fish species absences (17% of stream miles) with bank erosion the most pervasive physical habitat stressor (9% of stream miles). Acidity is also a widespread stressor (8% of stream miles affected), with acid deposition the most often diagnosed source of acidity. Urbanization and agricultural land use were also identified as important stressors responsible for fish species absences with both affecting more than 5% of stream miles.

Brook trout were most often absent due to physical habitat related stressors. Blacknose dace were rarely affected by urbanization or physical habitat; however, they appear to be affected by acidity. The bluespotted sunfish is one of the many fishes indigenous to the Coastal Plain that prefer naturally acidic streams; high pH was associated with many bluespotted sunfish absences. Liming associated with agriculture practices raises the pH of some naturally acidic streams (Figure 14-73), making them less suitable to acid endemic species (like blue-spotted and other species) and more suitable to other species (like blacknose dace that cannot tolerate low pH). Banded sunfish is another acid endemic species that prefers pH even lower than that for the bluespotted sunfish (Figure 14-74). The banded sunfish is typically associated with blackwater streams with slow water and naturally low acidity and high dissolved organic carbon. Other species including mud sunfish, ironcolor shiner, swamp darter, eastern mudminnow, pirate perch, creek chubsucker, tadpole madtom, redfin pickerel, as well as certain amphibian species including carpenter frogs, are

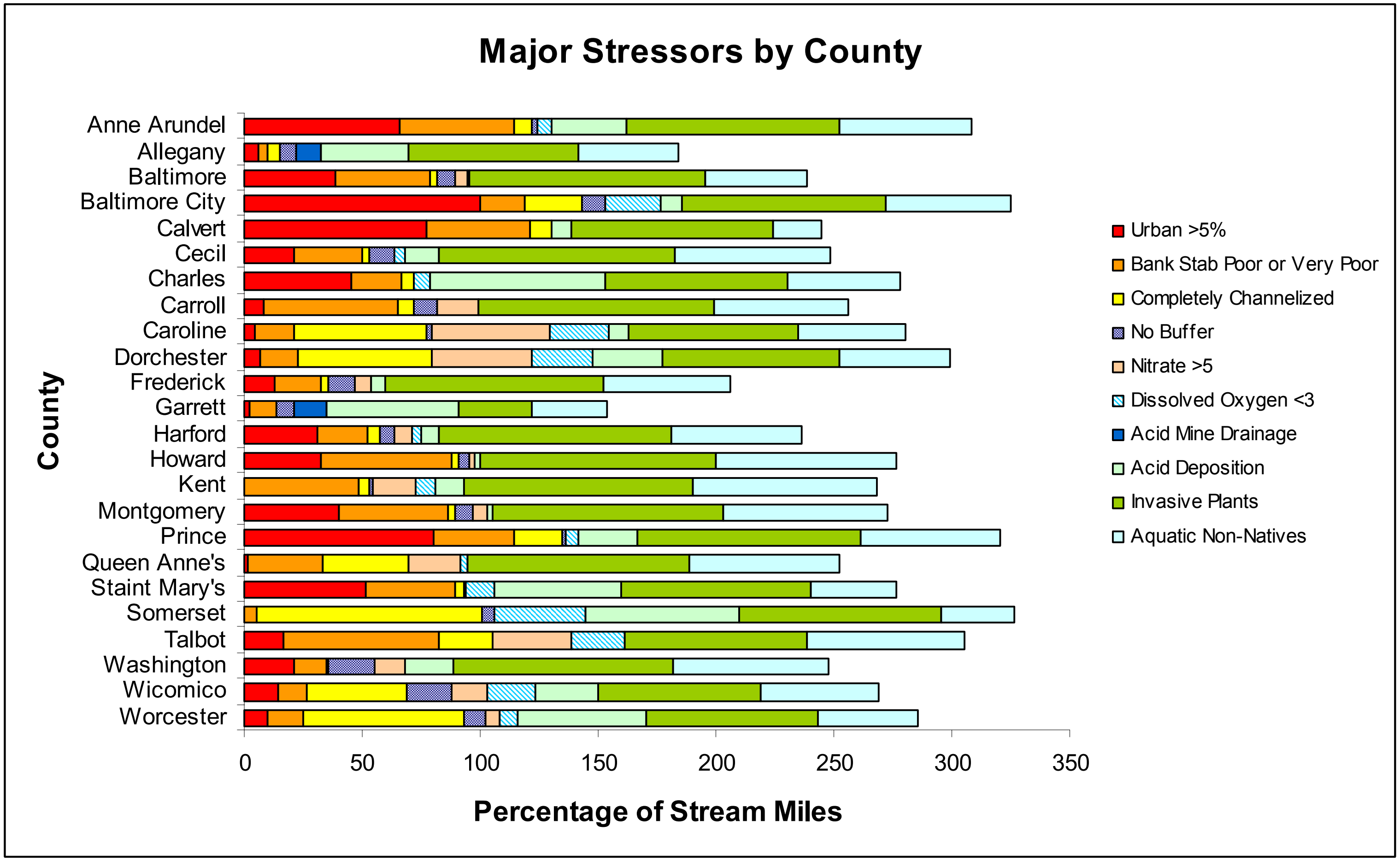


Figure 14-72. Stressors by county, in percent of total stream miles for that county, based on the 2000-2004 MBSS.

Table 14-16. Major stressors resulting in the loss of fish species (by all fish and by three selected species, bluespotted sunfish, blacknose dace, and brook trout) with number of stream miles affected statewide

Stressors (All species)	**% Stream miles (Std. error)**
Urban/Impervious land cover	5.74 (0.68)
Acidity (low pH or ANC)	8.32 (0.94)
Acid deposition	6.88 (0.84)
Acid mine drainage	0.32 (0.18)
Agriculture	0.50 (0.21)
Unknown	0.64 (0.34)
Physical habitat	16.93 (1.35)
Instream habitat	1.53 (0.45)
Epifaunal substrate	0.02 (0.02)
Velocity/depth diversity	1.03 (0.3)
Insufficient depth	2.24 (0.55)
Loss of canopy shading	1.51 (0.42)
Riffle embeddedness	0.52 (0.27)
Poor pool habitat	1.6 (0.48)
Poor riffle habitat	0.32 (0.23)
Channelization	2.32 (0.53)
Bank erosion	8.96 (0.99)
Agricultural land use	5.01 (0.78)
Nutrients (nitrate-nitrogen)	2.15 (0.49)
Low dissolved oxygen	2.45 (0.6)
Non-native salmonids	0.2 (0.14)
Stressors (Bluespotted sunfish)	**% Stream miles (Std. error)**
Acidity (high pH or ANC)	7.83 (1.67)
Physical habitat	2.25 (1.43)
Bank erosion	2.25 (1.43)
Forest loss	8.38 (1.59)
Agricultural land use	9.42 (1.8)
Sulfate (SO_4)	5.06 (2.13)
Low dissolved oxygen	1.8 (1.29)
Stressors (Blacknose dace)	**% Stream miles (Std. error)**
Urban/Impervious land cover	0.14 (0.1)
Acidity (low pH or ANC)	3.65 (0.72)
Acid deposition	3.22 (0.68)
Acid mine drainage	0.32 (0.21)
Agriculture	none
Unknown	0.11 (0.11)
Physical habitat	1.44 (0.44)
Velocity/depth diversity	0.07 (0.07)
Insufficient depth	0.17 (0.17)
Poor pool habitat	0.22 (0.16)
Channelization	0.39 (0.2)
Bank erosion	0.88 (0.35)
Forest loss	0.25 (0.25)

Table 14-16. (Continued)	
Stressors (Brook trout)	**% Stream miles (Std. error)**
Urban/Impervious land cover	9.44 (1.6)
Acidity (high pH or ANC)	3.79 (1.74)
Acidity (low pH or ANC)	1.79 (0.86)
Acid deposition	0.92 (0.6)
Acid mine drainage	0.98 (0.69)
Agriculture	none
Unknown	0.35 (0.35)
Temperature	3.45 (1.52)
Physical habitat	10.3 (2.35)
Instream habitat	1.97 (1.3)
Velocity/depth diversity	
Insufficient depth	0.37 (0.37)
Lack of canopy shading	2.99 (1.52)
Riffle embeddedness	1.97 (1.3)
Poor pool habitat	0.37 (0.37)
Bank erosion	7.66 (2.03)
Forest loss	5.44 (1.18)
Agricultural land use	1.93 (0.92)
Nutrients (nitrate-nitrogen)	7.09 (1.79)
Low dissolved oxygen	1.97 (1.3)
Non-native salmonids	1.65 (1.23)

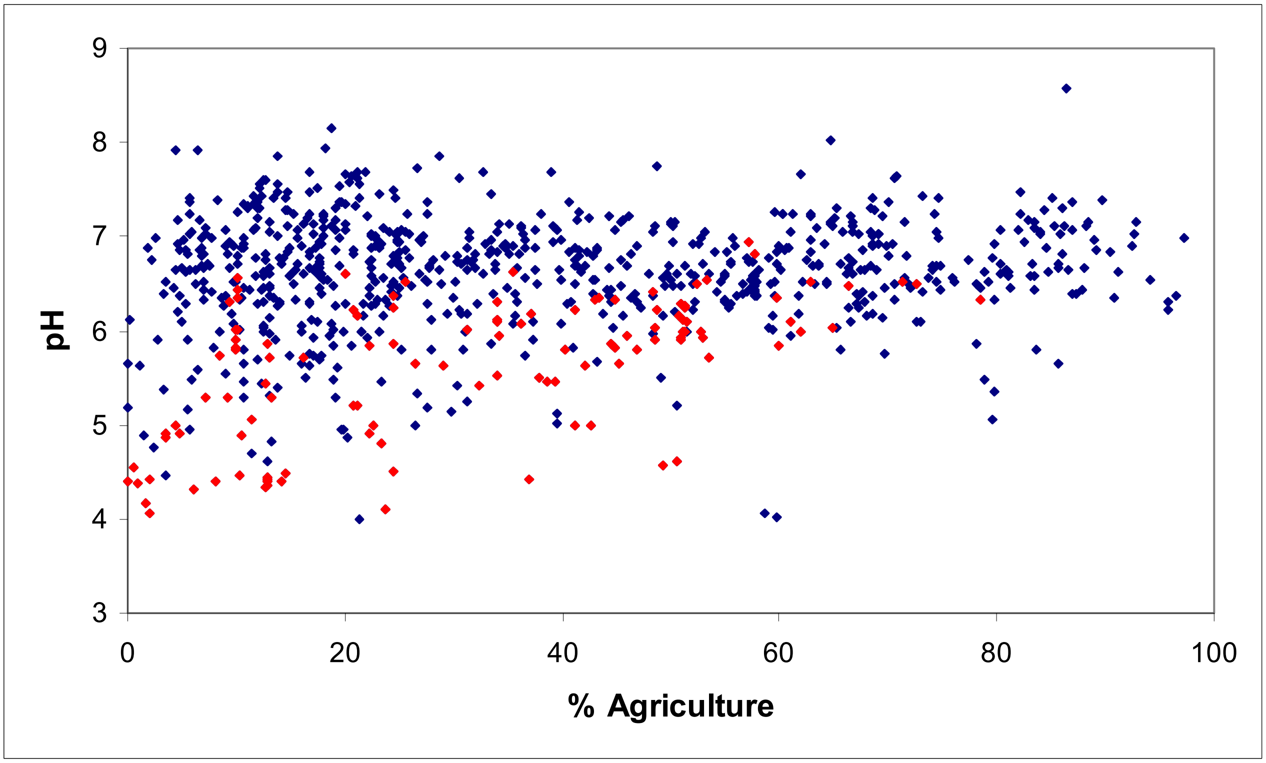


Figure 14-73. Relationship between the pH and the percentage of agricultural land use, statewide for 1995-2004 MBSS. Blackwater streams are indicated by red diamonds.

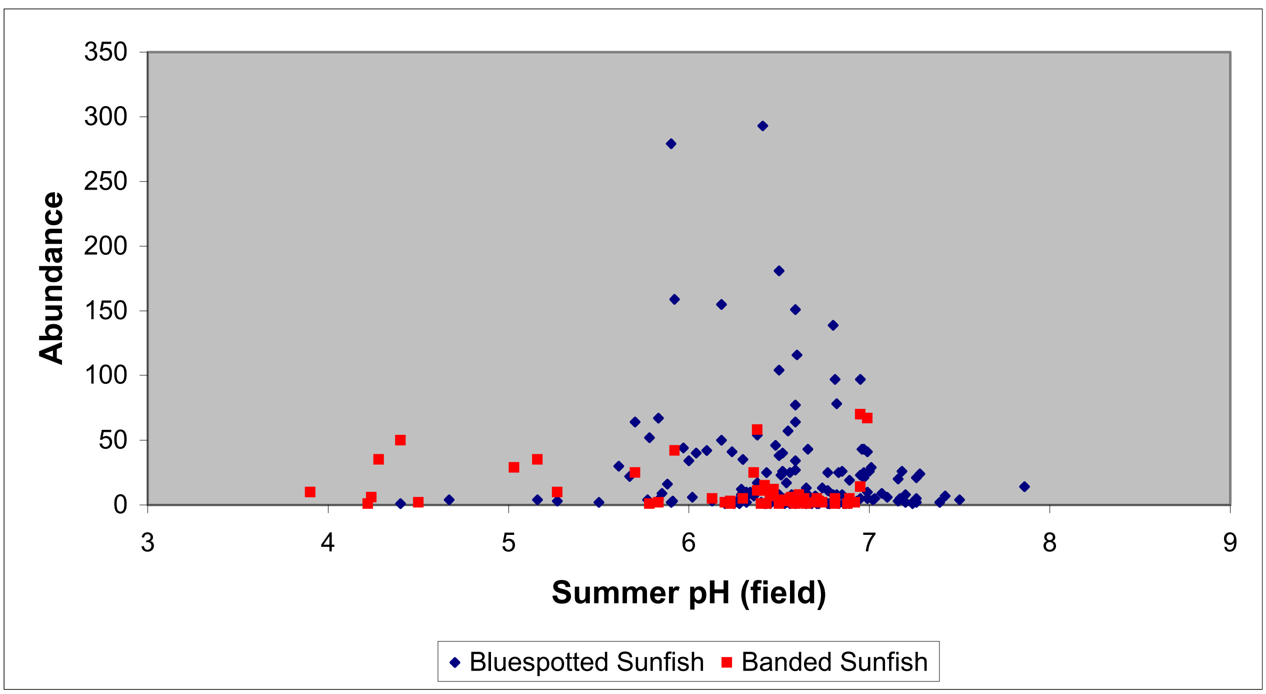


Figure 14-74. Relationship between the bluespotted sunfish and banded sunfish abundance and the summer pH, statewide for 1995-2004 MBSS. Bluespotted sunfish are indicated by black diamonds and the banded sunfish by red squares.

STRESSORS CAUSING FISHLESS STREAMS

The Prediction and Diagnosis Model (PDM) of Stranko et al. (2005a) identifies stressors causing the loss of certain fish species. What are the stressors present in streams with no fish? Based on the MBSS 2000-2004 data, 143 sites had no fish. Of these, 30% had values above the level of concern for one or more of eight stressors. Of these 143 fishless sites, 81 drained catchments of < 300 acres and only 15% were stressed, indicating that smaller streams naturally have fewer fish. In streams draining more than 300 acres, 45% were stressed. The list of stressors (i.e., human-related variables with values above the level associated with adverse effects) found at these fishless sites included the following:

- Channelization = 14 sites
- pH = 13 sites
- DO = 13 sites
- ANC = 11 sites
- Instream Habitat = 10 sites
- % Urban = 8 sites
- Bank Stability = 6 sites
- NO_3 = 0 sites

associated with relatively low pH. This indicates that both high acidity and pH resulting from agricultural liming must be considered in planning the protection of Coastal Plain biodiversity.

14.8 EXAMPLES OF STRESSOR IDENTIFICATION AT THE SITE AND BASIN LEVEL

Stressor identification at state and regional levels is critical to setting priorities and developing government strategies for stream restoration and protection. Specific restorations, however, require that stressors be identified at the scale of the restoration project, i.e., the site or the basin. While definitive stressor identification can require custom monitoring of the site or basin in question, MBSS data are often sufficient to characterize the major stressors of concern.

Volumes 1 through 5 describe that annual sampling of the MBSS and characterize the status of all Maryland 8-digit basins (although the smallest are combined into "super-basins" for assessment). Included in these characterizations are detailed tables listing the land use, water chemistry, and physical habitat values for each MBSS site. Using established thresholds of concern for many of these variables (e.g., ANC < 200 μeq/L or impervious surface > =10%), values exceeding the thresholds are flagged to indicate the likely presence of this stressor. The combined evidence from each of these tables is summarized in the "Interpretation of Basin Condition."

Biological data can also be used for diagnosing stream problems. The use of discernable patterns in the response of aquatic communities to stressors was first described as "biological response signatures" using Ohio EPA data by Yoder (1991) and has since been summarized in Simon (2003). The Ohio EPA approach used graphical techniques to describe fish and macroinvertebrate community responses to nine impact types: complex toxic, conventional municipal/industrial, combined sewer overflows/urban, channelization, agricultural nonpoint, flow alteration, impoundment, combined sewer overflows/urban with toxics, and livestock access (Yoder and Rankin 1995). Specifically, they identified metrics from the fish Index of Biotic Integrity, modified Index of Well-Being, and Invertebrate Community Index (e.g., darter species, percent round-bodied suckers, intolerant species, and percent Deformities, Erosions, Lesions/Ulcers and Tumors (DELT) anomalies) that were characteristic of each impact type. While percent DELT anomalies for fish and percent Cricotopus spp. for macroinvertebrates were indicative of the complex toxic impact types, and number of sunfish species, percent caddisflies, and qualitative EPS indicated the channelization impact type fairly well, there was much broader overlap among the other impact types.

The MBSS has developed a conceptually different, but very effective method for using fish data to identify likely stressors at individual MBSS sites—the Prediction and Diagnostic Model of Stranko et al. (2005a). This approach is based on determining which stream habitat conditions are suitable and preferred for individual fish species in the absence of anthropogenic stress.

Generating predictions of species presence. The first step in developing predictions for the model was to determine the physiographic provinces and basins where each species was collected in the development data set. The next step was to determine the stream conditions that each fish species prefers to inhabit, based on relationships of species abundance compared to nine variables that are not typically affected by anthropogenic influences to streams. These nine variables are referred to as predictor variables. The range of conditions that each species prefers to inhabit was determined for each predictor variable. Any value for a predictor variable that coincided with greater than average species abundance was considered to represent preferred conditions. The range of preferred conditions was considered to be the range of predictor variable values from the minimum to the maximum where each species was collected at greater than average abundance.

Once the ranges of preferred conditions were generated, they were used to generate predictions of the fish species expected to occur in a stream in the absence of major anthropogenic disturbance. Predictions were generated for the sites in the test data set using a hierarchical screening method (Smith and Powell 1971). Using this approach, a fish species was expected to occur at any site in Maryland that was in a physiographic province and basin where the fish species could be found and all of the predictor variables were in the preferred range for the species (in the development data set).

The expected list of species collected at each site in the test data set was then compared to the actual list of species collected. The species that were predicted to occur, but were not collected, were considered to be absent most likely due to the influence of anthropogenic stressors.

Stressor diagnosis. The first step in diagnosing probable stressors to fish species that were expected to occur, but were absent, was to determine the tolerance thresholds of each fish species to 14 stressor variables in the development data set. The tolerance thresholds for each species were considered the minimum and maximum value for each variable where the species was collected in the development data set. If the value for one of these stressor variables exceeded the tolerance thresholds for a species that was absent, but was expected to occur at a site in the test data set, then that variable was considered to be a stressor to the species. For ease of reporting, scores for the five habitat metric variables were combined into a single physical habitat structure stressor and pH and acid neutralizing capacity (ANC) were combined into a single acidity stressor. These combinations resulted in a total of nine stressors being reported.

14.8.1 Basin Examples Using the Fish Prediction and Diagnosis Model

In 1998, Maryland's basins most in need of restoration and protection were prioritized based on MBSS and other data as part of the state's Unified Basin Assessment (COMAR 1995-26.08.02.03). To assist in distributing limited funds within these priority basins, finer scale stressor information was needed to focus on specific areas where restoration and protection activities could be implemented. Based on MBSS data, sites affected by specific stressors were identified within priority basins.

The range of stress to MBSS sites from urbanization (as measured by percent impervious land cover) in the Gwynns Falls basin is shown in Figure 14-75. The southeastern portion of the basin is located in and near Baltimore City and is strongly affected by urbanization. In contrast, the northwestern portion of the basin is minimally affected.

Nitrate-nitrogen values in the Port Tobacco River basin are shown in Figure 14-76. One unnamed tributary suffers from elevated nitrate-nitrogen values. This stream originates in the town of La Plata and flows westward toward the Port Tobacco River. Other sites in the basin did not have elevated nitrate-nitrogen.

14.8.2 Site-Specific Examples Using the Fish Prediction and Diagnosis Model

Ultimately, site-specific restoration and protection of biota can only be achieved if the specific stressors affecting biota are known. As described above, the Prediction and Diagnosis Model (Stranko et al. 2005a) uses fish species tolerance thresholds to 14 stressor variables and nine non-stressor variables as determined from MBSS data. Non-stressor variable thresholds along with zoogeographic information were used to generate a list of the species expected to occur in a given stream in the absence of severe anthropogenic stress. Stressor variable thresholds were then used to diagnose probable stressors to fish species at sites where they were expected to occur, but were absent. The thresholds for the stressor variables diagnosed by the PDM can also be used to set minimum fish restoration and recovery endpoints. Stressor thresholds for species that are known to occur in a stream can also be used to set limits necessary to protect those species.

Table 14-17 shows results from five sites that illustrate a continuum of stream quality from severely degraded (none of the predicted species present) to minimally-degraded (all of the predicted species present). Probable stressors are listed, by species, for each example site.

Figure 14-77 shows the temperature value measured at the Carroll Branch site (26 ºC), from Table 14-17, and the maximum temperature threshold for brook trout (22 ºC). This illustrates how much one stressor would need to be reduced to result in suitable conditions for one species (brook trout). This is only one of potentially many stressors at this site. In addition, this value is based on a one-time reading of temperature and additional

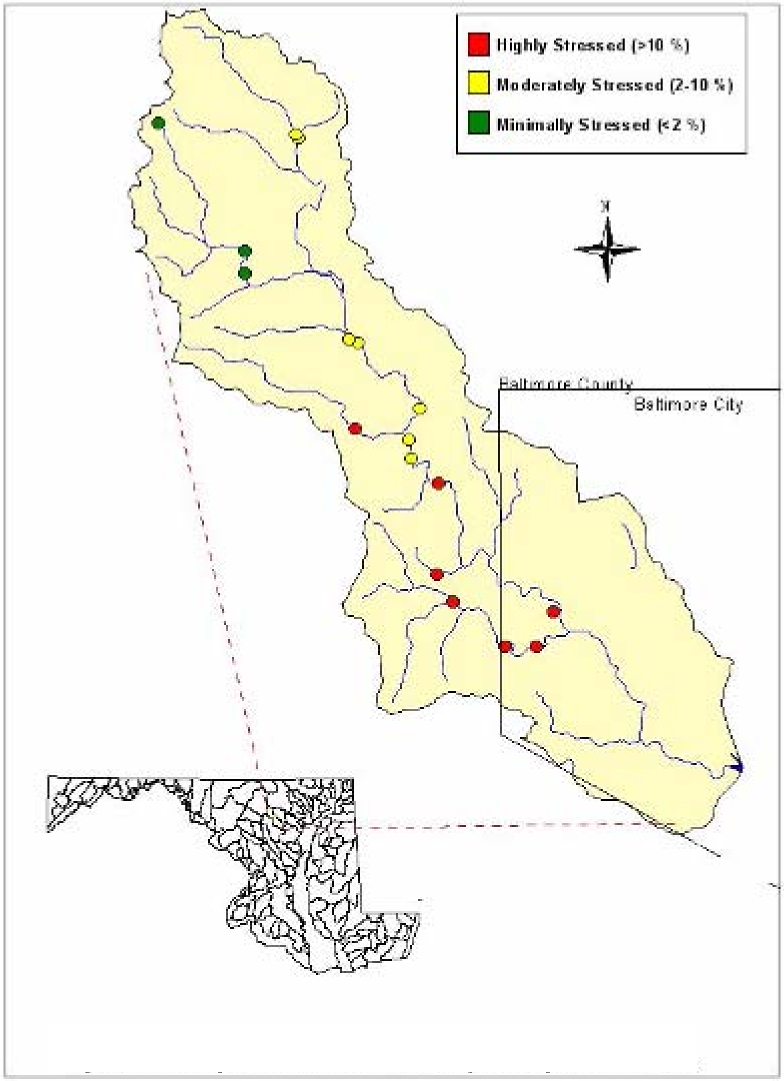


Figure 14-75. Map showing percentage of impervious surface in Gwynns Falls basin.

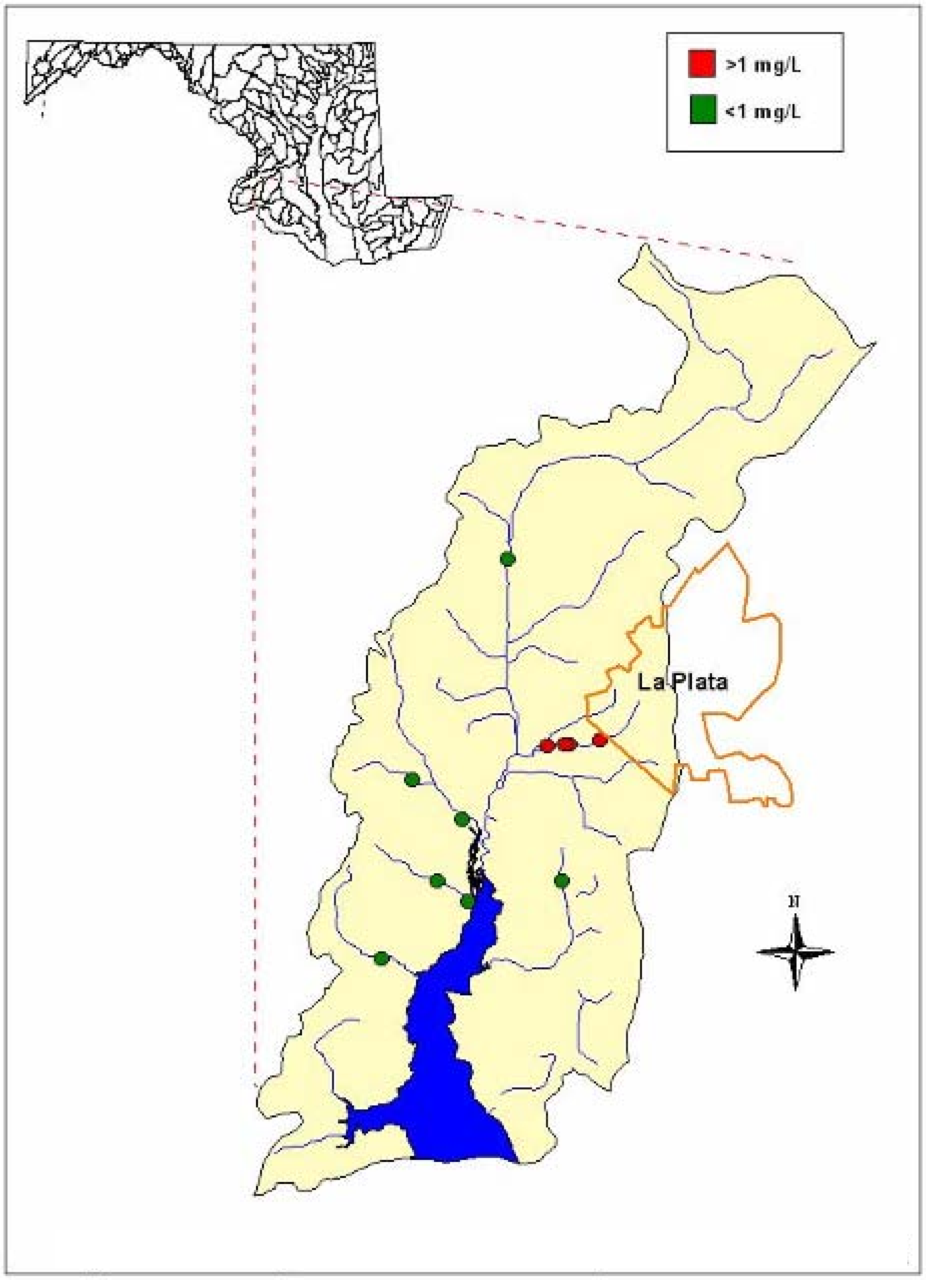


Figure 14-76. Nitrate-nitrogen concentrations at MBSS sites sampled in Port Tobacco River basin.

Table 14-17. Results from the Prediction and Diagnostic Model (Stranko et al. 2005) as applied to five sites to show a gradient of stream quality from severely degraded to minimally degraded. Probable stressors are those variables that exceeded tolerance thresholds for the species that was predicted but absent.

Stony Run, Baltimore City (0% of predicted species present).				
Species Predicted	**Species Present**	**Probable Stressors by species**		
Creek Chub		urban/impervious		
Rosyside Dace		urban/impervious	poor instream habitat	poor velocity/depth/diversity
Tessellated Darter		urban/impervious	poor instream habitat	
Woodland Creek, Kent County (29% of predicted species present)				
Species Predicted	**Species Present**	**Probable Stressors by species**		
Margined Madtom		agriculture land use		
Rosyside Dace		agriculture land use	nitrate/nitrogen	
Sea Lamprey		agriculture land use	nitrate/nitrogen	
White Sucker				
Redbreast Sunfish				
American Eel	American Eel			
Tessellated darter	Tessellated Darter			
Carroll Branch Tributary, Baltimore County (50% of predicted species present)				
Species Predicted	**Species Present**	**Probable Stressors by species**		
Brook Trout		temperature	agriculture land use	
Tessellated Darter			agriculture land use	
Blacknose Dace	Blacknose Dace			
Creek Chub	Creek Chub			
Wild Cat Branch, Montgomery County (90% of predicted species present)				
Species Predicted	**Species Present**	**Probable Stressors by species**		
Central Stoneroller				
Blacknose Dace	Blacknose Dace			
Creek Chub	Creek Chub			
Fantail Darter	Fantail Darter			
Longnose Dace	Longnose Dace			
Blueridge Sculpin	Blueridge Sculpin			
Potomac Sculpin	Potomac Sculpin			
Rosyside Dace	Rosyside Dace			
White Sucker	White Sucker			

Table 14-17. (Continued).		
Principio Creek Tributary, Cecil County, Sentinel Site (100% of predicted species present)		
Species Predicted	**Species Present**	**Probable Stressors by species**
American Eel	American Eel	
Blacknose Dace	Blacknose Dace	
Creek Chub	Creek Chub	
Blueridge Sculpin	Blueridge Sculpin	
Rosyside Dace	Rosyside Dace	
Tessellated Darter	Tessellated Darter	
White Sucker	White Sucker	

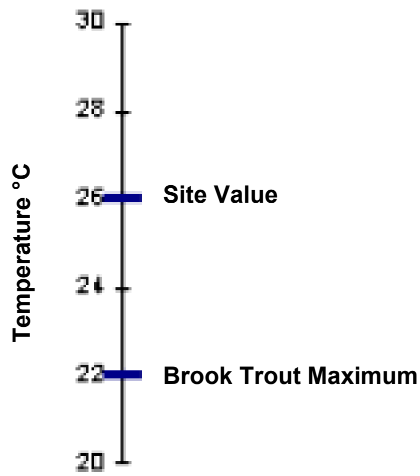


Figure 14-77. Comparison of temperature measured at Carroll Branch site and the maximum temperature threshold for brook trout.

measurements at this site would likely reveal even higher readings.

Figure 14-78 shows the percentage of impervious land cover in the catchment draining to the MBSS Sentinel site on a tributary to Principio Creek. The maximum tolerance thresholds to impervious land cover are shown for each of the species that were present at this site. In this relatively small (726 acre) basin, 400 acres of low-density residential development (0.5 to 5 acre plots) or 130 acres of high density residential (8 dwelling units per acre) or commercial land would likely result in a percentage of impervious land cover that exceeds the maximum threshold for the blueridge sculpin at this site. According to the thresholds for impervious land cover, as documented from MBSS, if the majority of the basin were converted to low-density residential land use, the entire assemblage would likely be lost. A much lower amount of high-density residential or commercial development (300 acres) would eliminate all of the species from this stream.

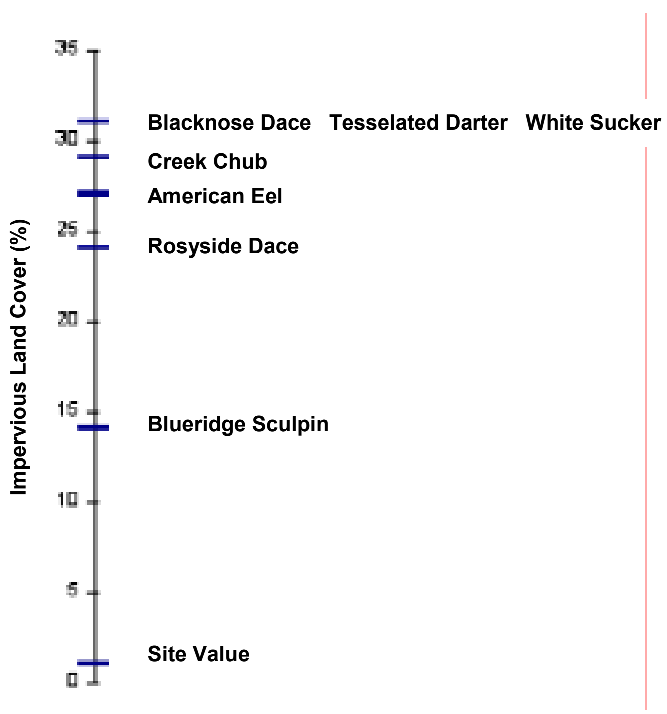


Figure 14-78. Comparison of the percentage of impervious land cover at Sentinel site on a tributary to Principio Creek and maximum tolerance thresholds of species present.

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